DISSERTATION

LAND USE EFFECTS ON PHYSICAL HABITAT AND NITRATE UPTAKE IN SMALL STREAMS OF THE CENTRAL ROCKY MOUNTAIN REGION

Submitted by

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WE HEREBY RECOMMEND THAT THE DISSERTATION PREPARED UNDER OUR SUPERVISION BY DANIEL WAYNE BAKER ENTITLED LAND USE EFFECTS ON PHYSICAL HABITAT AND NITRATE UPTAKE IN SMALL STREAMS OF THE CENTRAL ROCKY MOUNTAIN REGION BE ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY.

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ABSTRACT OF DISSERTATION

LAND USE EFFECTS ON PHYSICAL HABITAT AND NITRATE UPTAKE IN SMALL STREAMS OF THE CENTRAL ROCKY MOUNTAIN REGION

Watershed land use alteration and flow extraction influence the physical habitat and geochemical functions of small streams. In two associated studies, I explore stream physical habitat characteristics and nutrient uptake across a range of land use influences (flow extraction, agriculture, and urbanization). In the first study, I examined the effects of flow diversion to fine sediment deposition in a detailed field analysis pairing reaches above and below diversion dams on 13 mountain streams throughout north-central Colorado and southern Wyoming. Diversions are ubiquitous across the American West, yet previous studies on the impact of flow extraction have yielded mixed results. Through application of strict site selection criteria, multiple fine sediment measures, and an intensive sampling scheme, this study found that channels downstream of diversions contained significantly more fine sediment and slow flowing habitat as compared to upstream control reaches. Susceptibility to fine sediment accumulation was associated with decreasing basin size, bankfull depth, and d_{84}, and appears to be magnified in streams of less than 3% slope. In the second study, I investigate physical and hydraulic influences on transient storage and nutrient uptake in small agricultural and urban streams across a gradient of channel conditions and management modifications. Three geomorphically distinct segments on each of two streams were studied in the summer of

2007: one in a Colorado Front Range urban setting and the other in a mountainous agricultural region in north-central Colorado. The urban stream exhibits various levels of stabilization and planform alteration, and the agricultural stream has been subject to historically variable cattle-grazing practices. Reach-scale geomorphic complexity was characterized using highly detailed surveys of channel morphology, substrate, hydraulics, and habitat units. Injections of conservative bromide (Br^-) and non-conservative nitrate (NO_3^-) tracers were used to characterize channel processes. Geomorphic characteristics, specifically increased longitudinal roughness and flow depth, were strongly associated with both nutrient uptake and transient storage. Collectively, the studies underscore the primary influence of flow regime on habitat response and nutrient spiraling functions in the context of human influences.

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CHAPTER 1

INTRODUCTION

Streamflow is a master variable governing channel form, water quality, and ecological function. Altering any aspect of the magnitude, frequency, timing, duration, or rate of change of flow can directly influence physical and biological stream processes (Poff *et al.* 1997). Often flow-regime alteration is spurred by land use change, such as urban and agricultural development, including extraction of flows for consumptive use. Flow extraction leaves some channels with greatly reduced flow while trans-basin augmentation enlarges others. Development of urban and agricultural land alters the basin hydrologic and sediment budgets, at times leading to increased flow levels, flashier flow regimes, channel instability, and a loss of geomorphic complexity (Walsh *et al.* 2005). Such geomorphic responses alter flow depths, velocities, sediment patterns, and surface/groundwater interactions, ultimately affecting benthic habitat, transient storage, and nutrient spiraling processes (Houser *et al.* 2006, Jacobson *et al.* 2001). Alteration of stream sediment regimes is often blamed on upland erosion, but changes in runoff delivery may increase channel erosion, leading to downstream sedimentation. Besides indirectly altering channel inputs, direct channel modification can further impair stream function. Encroachments by urban and industrial development constrict river planform, resulting in the loss of floodplain connectivity and stream-channel homogenization.

Diversion dams and reservoirs alter sediment-transport continuity, and grade and bank control structures limit natural channel-adjustment processes (Ligon *et al.* 1995).

Managing flow and geomorphic interactions is crucial for the conservation and restoration of our nation's streams, but there is more to learn about the influences of channel and flow alteration on vital ecological functions, including hyporheic exchange and nutrient processing. Current stream-restoration practices are often narrowly focused on building static habitat features to restore channel stability, but the restoration of channel form does not imply return of comparable ecological function (Wohl 2005). A goal of this research is to improve management and restoration of the nutrient spiraling and geomorphological processes within streams through a better understanding of relationships among flow regimes, physical habitat templates, and ecological functions.

This dissertation describes two companion studies examining the effects of alterations in land use, channel form, and flow extraction on small urban, agricultural, and mountain streams. In Chapter 2, I describe an investigation of the geomorphic and ecological affects of stream flow alteration by diversion dams by contrasting the physical and hydraulic characteristics above and below diversion dams on headwater streams in the Rocky Mountains of north-central Colorado and southern Wyoming. Multiple sediment measures are explored to help elucidate the types of shifts in fine sediment, and susceptibility to fine sediment degradation is investigated with respect to channel and basin characteristics. Of the limited studies on the impacts of diversion dams on mountain streams, few have detected significant downstream alteration (Ryan 1997, Wesche *et al.* 1988, Bohn & King 2000, Albano 2006), in part due to a tendency for diversion locations to coincide with stream type transitions. The present study employs strict site selection criteria and a detailed field protocol designed to overcome this challenge. Implications for operation and design of diversion structures are discussed. In Chapter 3, I present a second study focused on geomorphic complexity, which provides a physical template for hyporheic exchange and nutrient uptake processes. Specifically, I examine the geomorphic settings of multiple reaches of an agricultural and an urban stream using a detailed physical habitat analysis to describe various forms of geomorphic complexity with both existing measures and novel metrics developed for this study. Tracer injections and biogeochemical modeling are used to estimate transient storage and nitrate uptake in each reach. Statistical modeling is used to examine which hydraulic and complexity measures influence transient storage and nitrate uptake. Finally, I discuss channel alterations that could enhance transient storage capacity and nitrate uptake, thus reducing stream nitrogen loads.

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CHAPTER 2

DOWNSTREAM EFFECTS OF DIVERSION DAMS ON THE BENTHIC HABITAT OF ROCKY MOUNTAIN STREAMS

2.1 Abstract

Streamflow extraction has the potential to reduce the sediment transport capacity of downstream channels and lead to an accumulation of fine sediments and habitat degradation. To investigate, I examined the effects of flow diversion on fine sediment deposition in a detailed field analysis pairing reaches above and below diversion dams on 13 mountain streams throughout north-central Colorado and southern Wyoming. Diversions are ubiquitous across the American West, yet previous studies on the impact of flow extraction have yielded mixed results. Through application of strict site selection criteria, multiple fine sediment measures, and an intensive sampling scheme, this study found that channels downstream of diversions contained significantly more fine sediment and slow flowing habitat as compared to upstream control reaches. Susceptibility to fine sediment accumulation was associated with decreasing basin size, bankfull depth, and d_{84}, and appears to be magnified in streams of less than 3% slope.

2.2 Introduction

All watersheds in the contiguous United States larger than 2,000 km^2 are controlled at some level by dams, resulting in over 75,000 structures included in the U.S.

Army Corps of Engineers' (USACE) National Inventory of Dams (NID) (Graf 1999). In reality, the number of structures with the potential to alter downstream water and sediment regimes is much greater as over 2,000,000 structures are not counted in the NID census (Poff & Hart 2002), owing to their small size and limited storage capacity. In the semi-arid western United States, potential degradation of stream habitat associated with flow depletion by relatively small diversion structures is related to increasing demands on available water resources that are already allocated. Adding to the challenge, the spatial and temporal distributions of water demand are often inconsistent with natural fresh water supplies. For example, in Colorado the majority of precipitation falls on the western side of the Continental Divide, while 61% of consumptive use takes place on the eastern side of the divide, requiring 24 trans-mountain water diversions across the Continental Divide (Litke & Appel 1989). There is also a seasonal discrepancy, with the majority of runoff occurring during the spring snowmelt, and the peak water use falls in late summer to early autumn, requiring over 12,750 reservoirs and 56,000 active points of diversion in Colorado alone (*Colorado's Decision Support Systems* (CDSS) 2007).

Stream flow diversions can exacerbate low flow conditions and produce extended drought-like periods, with lower flow volume and velocity, higher water temperatures, and less flow connectivity leading to the reduction of habitat area and quality (Miller *et al.* 2007). When streams are completely dewatered, the return of displaced macroinvertebrates often lags the return of flow to the channel (Boulton 2003). According to the U. S. Environmental Protection Agency's (USEPA 2006) *Wadeable Streams Assessment*, 36% of surveyed streams in the western United States suffer from poor or fair substrate condition due to accumulation of fine sediments. Fine sediment degrades macroinvertebrate habitat and also reduces primary productivity, species diversity, and abundance (Van Nieuwenhuyse & LaPerriere 1986, Lemly 1982, Wood & Armitage 1997, Waters 1995). Additionally, fine sediments choke salmonid spawning beds and interfere with egg incubation and fry emergence (Kondolf 2000) and decrease juvenile salmonid survival rates (Suttle 2004).

Flow extraction has a direct effect on stream ecological processes by altering the near-bed environment where macroinvertebrates reside and the disturbance regime of the channel (Statzner *et al.* 1988, Hart & Finelli 1999). Albano (2006) found severe reductions in macroinvertebrate densities when flows were reduced below 33% of ambient conditions. Rader & Belish (1999) found that the extent and timing of flow diversion appeared to affect the density and composition of macroinvertebrate assemblages in Colorado mountain streams. However, the factors of geomorphic setting and land use can either increase or decrease the overall effect of flow alteration by mediating the extent of changes. In highly diverted, small streams in New Zealand, greater amounts of coarse organic matter were retained and the wetted habitat area decreased, yet levels of conductivity, pH, and dissolved oxygen remained unchanged (Dewson *et al.* 2007). The biological implications of fine sediment and flow diversion have been well studied, but only a paucity of studies has assessed the direct physical and hydraulic effects of flow diversion. After discussing the available studies concerning physical processes in the next section, I outline my investigation of the downstream hydraulic and sedimentary effects of diversion dams on small streams, with particular focus on multiple measures of fine sediment.

2.3 Background

Quantifying changes in stream-sedimentation processes is complicated by high temporal and spatial variability and thus can be studied on multiple scales, from the entire basin to individual habitat patches. Sediment deposition is primarily activated by two processes: (1) a change in sediment loading, and/or (2) an alteration in stream hydrology. Sediment flux is a natural occurrence; however, land use changes can result in a direct alteration of both the quantity of available sediments and the flow available to transport them. In the Pacific Northwest, fine sediment appears to increase with basin disturbance across all lithology types, slope classes, and stream sizes (Kaufmann *et al.* 2009). Sediments in streams come from either the bed and banks of the channel and its tributaries or from the remainder of the basin, including upland hill slopes, agricultural fields, and urban development (Wood & Armitage 1997). Even in minimally disturbed basins, the washload (silt, clay, and fine sand particles) that is continuously delivered and transported through drainage networks across a wide range of flows provides a continuous source of sedimentation potential (Gordon *et al.* 2004).

An increase in sediment supply has been shown to homogenize bed texture, decrease average particle size, and diminish geomorphic variability (Buffington & Montgomery 1999, Bartley & Rutherfurd 2005). Laboratory flume studies modeling the effect of floodwater extraction on gravel channels demonstrate progressive increases in surface fines content as thalweg variability decreases at higher levels of flow extraction (Parker *et al.* 2003), and reduced hydraulic conductivity through substrate clogging that is only flushed away at moderately large floods (Schalchli 1992). Fine sediments also alter bed mobility, by increasing transport with greater levels of sand content (Wilcock 2001, Jackson & Beschta 1984).

In mountain streams, flow depletion can fundamentally alter channel hydraulics by increasing relative roughness and slow flowing habitat, thereby limiting fine sediment transport. A study examining channel dimensional and hydraulic properties above and below diversions in southern Wyoming and northern Colorado determined that the downstream channels of low-gradient streams (< 1.5%) were susceptible to channel alteration, but steeper channels were not (Wesche *et al.* 1988). Bohn and King (2000) made a concerted effort to select study sites with minimal variation in slope and valley confinement across the diversion, but found no correlation between stream gradient and channel change and only subtle decreases in channel conveyance, a varied sediment response, and no effects on vegetation. In a study of nine diversions in upper Colorado River basin streams similar to those in this study, Ryan (1997) reported reduced channel widths in unconstrained valley bottoms below diversions, but underscored the resiliency of subalpine channels and the periodic flooding that limits channel response. These previous studies have had difficulty detecting the effects of diversion dams due to maintenance of downstream channel geometry by the passage of flood flows and the tendency for diversions to be located at breaks in gradient and geomorphic context.

2.4 Study Description and Objectives

In this study, I searched for a greater understanding of the downstream geomorphic influences of diversion dams. The specific objectives were to: (1) select diversion sites of similar channel characteristics between a reference reach above the diversion and a representative reach below the diversion to minimize inherent geomorphic differences prior to diversion construction; (2) quantify and compare a multitude of measured and computed channel characteristics above and below selected diversions; and (3) identify the predominant factors contributing to fine sediment accumulation in diverted streams. In the summer and fall of 2005, I quantified the effects of agricultural and municipal diversion from 13 Rocky Mountain streams. Using strict site selection criteria followed by detailed quantification of the geometric, hydraulic, and sedimentary characteristics both above and below diversion dams, I assessed the effects of diversion structures and flow extraction on the physical and ecological conditions of diverted streams. Accordingly, I focused on four general hypotheses:

1. The channels selected for this study have no significant differences in channel slope or average substrate size between channel reaches above and below the diversion structures.
2. There are significant differences between the dimensional, sediment, hydraulic, and habitat characteristics of the upstream reference reach versus downstream reach.
3. There exist statistically significant subsets of measured hydraulic and geomorphic factors that explain observed variability in fine sediment accumulation in both fast and slow flowing habitat patches.
4. The following settings lead to greater differences in accumulation of fine sediments due to flow diversion:
    a. lower-gradient channels,
    b. hydraulically rougher channels,

c. channels with blockages creating more backwater, or

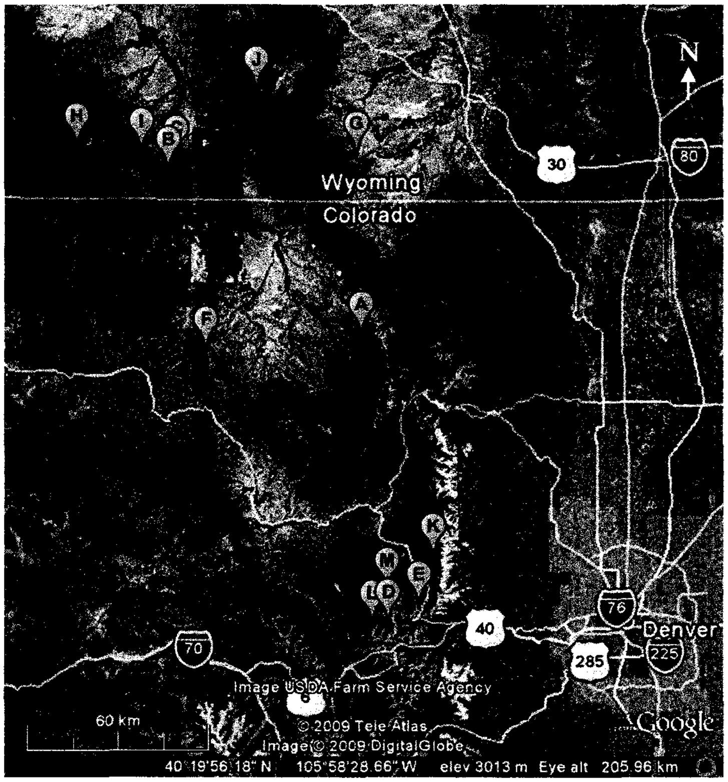
d. basins with a higher sediment supply.

2.5 Methods

The detailed field and laboratory methodology described below is designed to overcome some of the limitations of previous studies and highlight effects of diversion dams on stream hydraulic and physical conditions. The study design reflected a balance between achieving a sufficient number of field sites and adequate detail at each site. Detailed physical surveys of 26 paired reaches were ultimately performed above and below 13 diversion dams.

2.5.1 Study Design and Stream Site Selection

To study the effects of diversion dams on the physical conditions of small streams, upstream control reaches were compared to downstream diverted reaches at the 13 study sites in northern Colorado and southern Wyoming (Figure 2.1). All diversion dams were located on U. S. Forest Service land and were dispersed among the Williams Fork, Fraser, North Platte, Laramie, and Little Snake river basins. The construction and materials of the low-head diversion dams in this study varied widely, from seasonally constructed rock, wood, and tarp structures to permanent concrete dams with multiple gates and spillways. Most of the dams are primarily for agricultural use, but five of the diversions, in the Fraser River and Williams Fork River basins, are operated by Denver Water for municipal use.



Map Key	Site Code	Site Name
A	CAN	South Fork Canadian River
B	BCO	Billie Creek One
C	BCT	Billie Creek Two
D	BOB	Bobtail Creek
E	CUR	Current Creek
F	GRZ	Little Grizzly Creek
G	FOX	Fox Creek
H	HAG	Haggerty Creek
I	MIN	North Fork Miners Creek
J	NFR	North French Creek
K	RAN	Ranch Creek
L	SMN	Steelman Creek
M	STL	St. Louis Creek

Figure 2.1 – Map of stream diversion sites included in this study

Often diversion dams are constructed at points of change in valley confinement and slope, thus alterations due to the structure are often overshadowed by reach-scale differences in the stream prior to dam construction. Thus, it was critical to select sites with matching stream character across the diversion dam. Previous work analyzing the effects of flow diversions (Bohn & King 2000, Ryan 1997, Wesche *et al.* 1988) has underscored the dominance of antecedent stream and valley conditions and the need for careful selection of comparison reaches. Accordingly, available diversion records and area maps were used to identify candidate diversion sites in relatively unaltered basins. These basins (defined as the area upslope of the diversion dam) contained no flow extractions or augmentations and minimal anthropogenic land use activities. During on-site visits, the collective engineering and ecological judgment of the research team assessed the reach-scale similarities above and below each diversion. We focused on bed slope, channel planform, stream type, valley confinement, vegetative influences, and lithology. If we concluded that the study reaches varied fundamentally in these features before the construction of the diversion dam, the site was rejected. To examine the effects of diversion magnitude, we endeavored to select sites where diversions were likely to operate for the duration of the study season (early summer to mid fall) across a gradient of base flow diversion from minor to complete. Lack of control over the diversion operation and seasonal access restrictions prevented the research team from measuring all streams in both seasons. Initial field visits were performed on 11 streams in July on the lower end of the falling limb of the snowmelt hydrograph and final visits on 7 of the 11 original streams, plus two additional streams in September and October at base flow conditions, resulting in a total of 20 stream visits (Table 2.1).

As basins of minimal upstream anthropogenic alteration tend to be high in elevation in this region, selected diversion sites were all above 2300 m. The steep, clear-water study streams were characterized by high sediment transport capacity and comparatively low fine sediment supply from spruce-fir-dominated watersheds.

Table 2.1 – Study site elevation and sample times

Site Code	Site Name	Elevation (m)	Field Visits Summer	Fall
BCO	Billie Creek One	2747	x	
BCT	Billie Creek Two	2748	x	
BOB	Bobtail Creek	3185		x
CAN	South Fork Canadian River	2671	x	x
CUR	Current Creek	3463	x	x
FOX	Fox Creek	2299	x	
GRZ	Little Grizzly Creek	2595	x	x
HAG	Haggerty Creek	2574	x	x
MIN	North Fork Miners Creek	2557	x	x
NFR	North French Creek	2916	x	x
RAN	Ranch Creek	2894	x	
SMN	Steelman Creek	3184		x
STL	St. Louis Creek	2824	x	x

2.5.2 Reach Characterization

The essence of this study was to compare the reference upstream reach condition with the diverted downstream reach condition across all sites. Reach length was selected to be approximately 16 times the bankfull width of the upstream reference channel, to encompass greater than one meander wavelength of sinuous streams, and was divided into eight equally spaced transects.

2.5.2.1 Habitat Classification

Habitat variation within each of the 26 reaches was characterized by the linear distribution of habitat types along each of the eight transects. The Hawkins *et al.* (1993) hierarchical classification of sub-reach habitat types was used to first divide stream units into fast and slow water (Level I), then further parse fast water into turbulent versus non-turbulent and slow water into scour versus dammed pool (Level II). The Level II classification scheme was initially employed, but variation among study sites required simplification to Level I; thus the field recorded riffles, runs, and cascades were grouped into a "fast" category and pools were classed in the "slow" category. Specifically, any section of stream containing moderate to rapidly moving flow and surface disturbances was classified as a fast habitat zone, whereas slow habitat zones were demarcated by a smooth water surface and relatively slow flow velocity. The proportion of transect length each habitat unit occupied was then transformed to the proportion of that habitat for the entire reach. Some study reaches, namely plane bed streams that do not characteristically have pools, were identified by this protocol as having no slow flowing habitat. Yet, slow habitat sediment samples were taken from meter-scale slow patches along the margins of the channel.

2.5.2.2 Physical and Hydraulic Measurements

The various dimensional, hydraulic, and textural properties of each stream were measured at multiple scales. Large-scale (cross-section to reach) characteristics were recorded with physical measurements and habitat characterization, but smaller scale characteristics were analyzed using local streambed surface and subsurface sampling within fast and slow habitat units.

Each of the eight transects was measured for wetted and bankfull widths in addition to the depth, velocity, and location of the thalweg. The downstream profile was surveyed using an autolevel. A channel type classification for mountain streams by Montgomery and Buffington (1997) was used for classification of study reaches as it corresponded quite well to channel type sequences and slopes observed in the study domain. Additionally, two cross-sections, representative of the reach as a whole, were surveyed in detail to determine bankfull and wetted channel characteristics of each reach.

Discharge was measured using the velocity-area cross section method (Harrelson *et al.* 1994). Depth-averaged velocity was measured with a Marsh McBirney Flo-Mate™ portable flow meter on a calibrated wading rod at greater than 10 equally spaced points across the channel. The mean of two cross-sectional-averaged flows determined the reach-averaged value. Additionally, point measurements of flow depth and depth-averaged velocity were recorded at each substrate sampling site and at the thalweg of each cross-section.

The distribution of USGS streamflow gauges is disproportionately skewed toward larger streams and rivers (Poff *et al.* 2006). Locating small streams in Colorado and Wyoming with gauges both upstream and downstream of a diversion, that also met our selection criteria, proved infeasible for all sites. As such, time-series streamflow data were not available and we had to rely on measurements during field visits. Evidence from flow measurements during multiple visits to each site suggests that diversion rates did not vary substantially during the operation season, leading to an assumption that instantaneous measurements of flow conditions were sufficiently representative of average conditions over the duration of the study.

In the absence of historical data, I focused on three types of diversion measures including: (1) a fraction of flow per measured width of channel, (2) flow depth scaled a representative coarse sediment diameter, and (3) the flow relative to that of the control reach upstream. Unit discharge, which scales the cross-sectional discharge by the wetted width, effectively describes the volumetric flow per unit width of channel. Relative submergence is defined as the ratio of average flow depth to the 84th percentile sediment size (d_{84}); as flow depth increases the ratio increases into the range between 1 and 4, beyond which wave drag markedly decreases and particle roughness is minimal (Bathurst 2002). Percent diverted is a straightforward measure of the difference between the upstream and downstream flow. An extensive set of standard hydraulic descriptors (see Garcia 2008 and Gordon *et al.* 2004 for definitions) were calculated using the basic measurements of channel form and flow characteristics described above, and in some instances, pebble count data as described in the next section.

2.5.2.3 Bed Sediment Sampling

We developed a suite of three independent fine sediment measures with the aim of providing greater resolution than many standard protocols (*e.g.*, Faustini & Kaufmann 2007). Specifically, fine sediment, defined in this project as sediments less than 2 mm in diameter (including sand, silt, and clay), was measured using three intensive methods:

1. The surface sediment of each reach was quantified using a 400 point pebble count. Selected pebbles were distributed along the reach by taking 50 samples at each of the eight transects using a grid-style sampler (Bunte & Abt 2001).
2. Three points in the fast and slow habitat of each reach were selected for local cylinder sampling of fine sediment. The fast habitat samples were limited to areas where a steel cylinder could be driven into the bed and the slow habitat samples avoided heavy patches of algae and the channel margins. At each location a 0.25 m diameter steel cylinder was driven into the bed. Within the cylinder the bed was agitated and an aliquot sample was removed from the

water column. Then samples were collected to a depth of approximately 10 cm and field-separated into ±6 mm sizes. The coarser fraction was drip dried and weighed on-site and the finer fraction sieved in the laboratory at 5.6 mm, 2 mm, 0.5 mm, and 0.25 mm intervals. The portion finer than 2 mm plus the suspended solids from the aliquot sample were used to calculate the percent mass fines per specified volume of sediment. These measures will be reported below as "volumetric percent fines."

3. Finally, a local-scale surface presence/absence areal count of fine sediment was performed with the grid sampler proximal to all cylinder sampling locations. Fifty points were noted at each of the six sampling locations, for a total of 300 points per reach.

For comparison among sediment measures, all are expressed as the percent of fine sediment. The areal and pebble count measures are a percent of a total count of particles and the volumetric measure is a percent of mass extracted from the cylinder sample.

2.5.2.4 Geology and Sediment Supply

In addition to field observations of bed and bank stability at each study site, a larger spatial analysis of the lithology and sediment availability of the upstream drainage basins was performed by identifying the dominant underlying surface geology of the basins upstream of each diversion structure (Green 1992, Green & Drouillard 1994) using maps generated in ArcGIS™ 9 (Environmental Systems Research Institute, Inc. (ESRI), Redlands, California, USA). The predominant lithology of each largely undisturbed basin was then stratified based on its sedimentation potential (Reid and Dunne 1996).

2.5.2.5 Diversion Structures

The site-specific backwater extent and permeability of individual diversion dams could highly influence fine sediment and flood passage. Indeed, the construction and materials of diversion dams was observed to vary widely, from seasonally reconstructed rock, wood, and tarp structures to permanent concrete dams with multiple gates and spillways. To limit local effects, reaches were located a sufficient distance upstream and downstream of the diversion structure to eliminate any local backwater or scour effects. In addition, the likelihood of high flow passage and blockage of fine sediment at each structure were rated with four-tiered class variables.

2.5.3 Data Analysis and Statistics

Survey results for cross-sectional and longitudinal data were post-processed using Microsoft Excel® spreadsheets and Visual Basic® routines (Microsoft, Redmond, Washington, USA). Channel slope was calculated by fitting a linear trend line to the measured water surface elevations for each reach. Additionally, cumulative distribution functions of pebble counts were calculated and summarized as descriptors of sediment distribution (d_{16}, d_{50}, and d_{84}).

All parameters measured in the field as equal to zero (i.e. no fine sediment in a grid count, or zero velocity as read by the flow meter) were adjusted to one-half of the detection limit for analysis. Statistical calculations were performed using SAS® 9.2 (2008, SAS Institute, Inc., Cary, North Carolina, USA). Parameters for both above and below diversion channel characteristics were tested for normality. Due to the tendency for small sample sizes (n = 20) to pass parametric normality tests, analysis shifted to the more critical graphical evaluation of quantile-quantile plots and histograms. Investigation of these graphs of all 43 variables led to the conclusion that non-parametric statistical testing would be appropriate.

The upstream vs. downstream comparisons of the 20 site visits were evaluated with the non-parametric, one-tailed, Wilcoxon signed rank test at $\alpha = 0.10$ level. For *a posteriori* verification of channel similarity above vs. below each diversion dam, I analyzed the differences in the water surface slope and d_{50} between the upstream and downstream study reaches. Similarly, to test the hypotheses that diversion dams cause significant changes in channel physical and hydraulic characteristics between the upstream control reach and the downstream diverted reach, Wilcoxon signed rank tests of the differences in channel dimension, substrate, hydraulics, and habitat variables from below the diversion to above were performed. To investigate whether low-gradient streams are more susceptible to fine sediment accumulation when subject to flow extraction, the data set was split into two groups of ten site visits: < 3 % versus ≥ 3 %.

Multiple methods were examined to express the difference in fine sediment accumulation between the reference upstream and diverted downstream reaches. Preliminary regression analyses focused on fine sediment variables expressed as a percent change in sediment from above to below the structure. However, low values for percentage of fine sediment in the control reach at some sites resulted in a very small divisor in the percent change relationship. This numerical issue caused multiple order-of-magnitude differences in parameter values which led to extreme outliers for regression analysis. Instead, an arithmetic difference between the percent fines downstream and upstream of the diversion was used. It is acknowledged that the application of this simple difference does not normalize systems with greater or lesser overall amounts of fine sediment; hence two sites would have a 5% difference in fine sediment whether they had 30% upstream and 35% downstream or 1% upstream and 6% downstream..

Testing the hypothesis that hydraulic and geomorphic factors explain significant variance in fine sediment accumulation in both fast and slow flowing habitat patches required multiple steps. Forty-three variables were either directly measured in the field or calculated from field measured values; thus it was necessary to pare down candidate descriptors to those that contained unique, non-redundant information with a straightforward physical interpretation. First, principal components analysis (PCA), using SAS® 9.2 (2008), was used to reduce redundancy among parameters and extract the variables that contained the most unique information (Jolliffe 2002). A reduced set of orthogonal PCA axes did not lend itself to clear interpretation, thus field measured variables were preserved. Finally, eight variables were selected for best subsets regression analysis (SAS® 9.2, 2008) using the information from the PCA analysis and physical understanding as to which variables could provide information about susceptibility to fine sediment accumulation and be easily measured (Table 2.2). Subsets were sorted by Mallow's Cp ranking and the top ten models of two parameters or less were analyzed for patterns in the selection, significance, and response direction of repeating variables.

Table 2.2 – Variables included in best subsets analysis to predict changes in fine sediment levels across diversion structures

Variable	Units
Percent of flow diverted	%
Bankfull dimensionless shear stress	-
Drainage basin area	km^2
Unit discharge	m^2/s
Bankfull depth	m
Darcy's friction factor	-
84^{th} percentile grain size	mm

2.6 Results

2.6.1 Site Conditions

The 13 study sites (paired reaches) are located in mountainous terrain, exhibit snowmelt hydrology, and have gravel to cobble beds with d_{50} between 5 mm and 124 mm. Flow extraction during field visits ranged from 23% to 99% of the upstream flow and stream types ranged from pool riffle to step pool, with slopes between 1.3% and 15.7% (Appendix A).

2.6.2 Sediment, Hydraulic, and Channel Alteration

Even with stream type, slope, and channel character matched across the diversion sites, 33 of 43 measured or calculated variables differed significantly ($p < 0.10$) between the channel upstream of the diversion and the channel downstream of the diversion (Table 2.3).

Table 2.3 – Results from Wilcoxon signed rank test of differences between upstream reference reach and downstream diverted reach (bold values significant at a $p < 0.10$)

Variable	Units	Response Direction (Below – Above)	All Sites	Low Slope (<3%, n = 10)	High Slope (>3 %, n = 10)
Volumetric % Fines - Fast	-	+	**0.002**	**0.004**	0.242
Volumetric % Fines - Slow	-	+	0.763	0.625	0.432
Areal % Fines - Fast	-	+	**0.044**	0.193	0.131
Areal % Fines - Slow	-	+	**0.074**	0.160	0.232
Pebble Count % Fines*	-	+	**0.052**	**0.094**	0.438
Proportion Slow	-	+	**0.048**	0.383	**0.084**
Bed Slope*	m/m	–	0.588	0.563	0.297
Flow Rate	m^3/s	–	**< 0.001**	**0.002**	**0.002**
Wetted Width	m	–	**< 0.001**	**0.002**	**0.002**
Unit Discharge	m^2/s	–	**< 0.001**	**0.002**	**0.010**
Cross-sectional Area	m^2	–	**0.014**	0.106	**0.084**
Hydraulic Depth	m	–	0.210	0.570	0.232
Hydraulic Radius	m	–	**0.004**	0.106	**0.027**
Wetted Perimeter	m	–	**0.015**	**0.049**	0.131
Bankfull Width*	m	–	**0.011**	**0.063**	0.156
Bankfull Area*	m^2	–	**0.040**	**0.063**	0.297
Bankfull Depth*	m	–	0.216	0.313	0.469
Average Velocity	m/s	–	**< 0.001**	**0.002**	**0.027**
Shear Velocity	m/s	–	**0.003**	**0.065**	**0.027**
Shear Stress	N/m^2	–	**0.002**	**0.049**	**0.037**
Dimensionless Shear Stress	-	–	0.231	0.770	**0.065**
Particle Reynolds Number	-	–	**0.004**	**0.037**	**0.049**
Reynolds Number	-	–	**< 0.001**	**0.002**	**0.010**
Froude Number	-	–	**< 0.001**	**0.004**	**0.037**
Grain Froude Number	-	–	**0.004**	0.106	**0.020**
Average Stream Power	W/m	–	**< 0.001**	**0.002**	**0.002**
Unit Stream Power	W/m^2	–	**< 0.001**	**0.002**	**0.004**
Dimensionless Unit Stream Power	-	–	**0.047**	0.695	**0.010**
'Manning's Roughness	-	+	**0.001**	**0.020**	**0.049**
Average Thalweg Depth	m	–	**< 0.001**	**0.006**	**0.002**
Average Thalweg Velocity	m/s	–	**0.001**	**0.020**	**0.027**
Average Sediment Size*	Mm	–	0.497	0.438	0.938
84th Percentile Sediment Size*	Mm	–	0.340	0.844	0.375
Relative Submergence R/d_{50}	-	–	0.368	0.695	**0.065**
Relative Submergence R/d_{84}	-	–	**0.033**	0.322	**0.037**
Average Depth - Fast	m	–	**0.081**	0.680	**0.027**
Average Depth - Slow	m	–	0.504	0.443	0.752
Average Velocity - Fast	m/s	–	**0.015**	0.770	**0.002**
Average Velocity - Slow	m/s	–	**0.001**	**0.031**	**0.047**
Unit Discharge - Fast	m^2/s	–	**0.040**	0.846	**0.002**
Unit Discharge - Slow	m^2/s	–	**< 0.001**	**0.023**	**0.006**

* These variables were calculated with from site sample set of n = 13 due to the measurements being performed only once, not for each field visit

The subset of 10 channels with less than 3% slope contained significantly more downstream fine sediment by both the pebble count (p = 0.094) and volumetric fines in fast zone (p = 0.004) measurements, whereas the steeper channels ($\geq$ 3%) did not exhibit any significant differences in fine sediment.

There were significant differences in four of the five fine sediment metrics between upstream reference reaches and downstream diverted reaches, with the exception being volumetric fines in slow flowing zones (p = 0.763). Slow zone volumetric and areal samples had a significantly higher percentage of fines ($p < 0.001$) than fast zone samples and downstream diverted reaches had significantly more slow habitat (p = 0.048) than upstream reference reaches. Flow diversion also significantly decreased several hydraulic variables between the upstream and downstream channel, including flow velocity ($p < 0.001$), average shear stress (p = 0.002), and unit stream power ($p < 0.001$).

Channel slope (p = 0.54) and channel substrate (d_{50}) (p = 0.15) were not significantly different between the reaches above and below the diversion, confirming that paired sites were fundamentally similar prior to diversion construction. The independence of the 11 summer and 9 fall field visits (with 7 visited in both seasons) was tested using parametric mixed effects model to examine the viability of using a single dataset of n = 20. Three of the four local fine sediment measures had no seasonal effect, with only areal fines in the slow zones showing a marginal seasonal effect (p = 0.092). No site effect was detected in three of the four sediment measures with the exception of volumetric fines in the fast zones (p = 0.032). With limited seasonal and site effects, it was concluded that combining all site visits into a single dataset of n = 20 would be appropriate (P. Chapman, CSU Dept. of Statistics, pers. comm. 2009)

2.6.3 Regression Analysis

Predictors for change in areal fines between the downstream and upstream reaches were dominated by inverse relationships with upstream bankfull depth, basin size, and d_{84}. All sediment and habitat changes were calculated as the difference between downstream and upstream. Change in volumetric fines in fast flowing zones showed a strong negative relationship with d_{84} (Table 2.4). Change in volumetric fines in slow zones, not found to be significantly different across study sites in Wilcoxon signed rank tests, likewise had no significant regression models. A few weak relationships with change in volumetric fines in slow zones were initially reviewed, but removal of an outlier (CAN_1) eliminated all significant relationships. Change in pebble count fines had consistent inverse relationships with upstream bankfull depth. Finally, the percentage change of slow habitat showed a strong positive relationship with percent diverted. Shifts in relative submergence and unit discharge were also considered as measures of flow diversion, but neither proved significant in any regression models. Additionally power models were investigated but yielded no substantial improvements over linear models. The representative regression models (Table 2.4) are physically interpretable and contain variables that consistently explained the most variance in fine sediment and habitat response.

Table 2.4 – Representative regression models

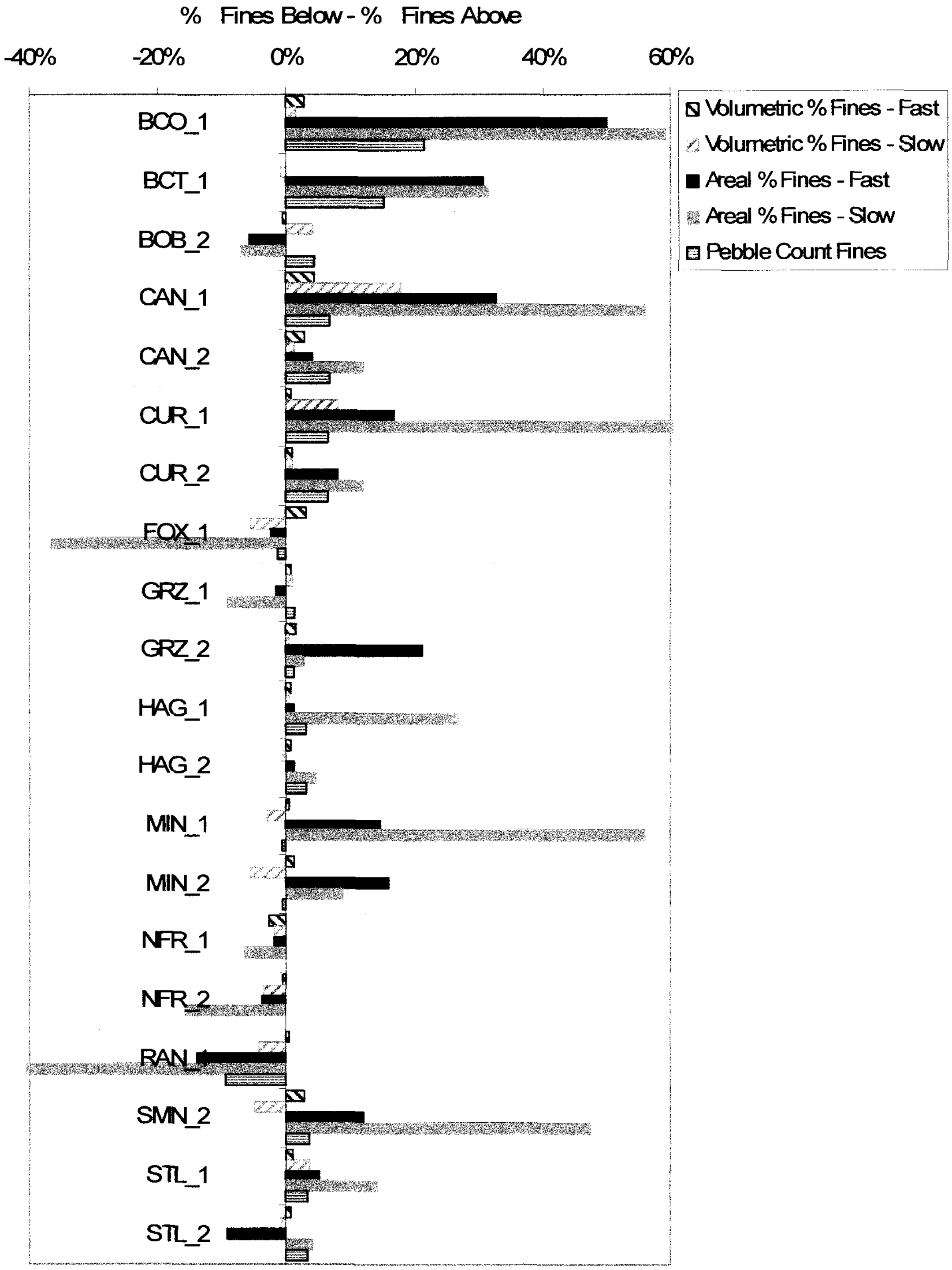
Dependent Variable	Representative Regression Models	Model p-value	Adjusted R^2	β_1 p-value	β_2 p-value
Change in Volumetric % Fines – Fast	0.0301 - 0.109*d_{84}	0.011	0.26	0.011	-
Change in Volumetric % Fines – Slow	no significant models				
Change in Areal % Fines – Fast	0.479 - 1.068*D_bf	<0.001	0.57	<0.001	-
Change in Areal % Fines – Slow	0.591 - 0.0089*Basin - 1.647*d_{84}	0.006	0.39	0.005	0.034
Change in Pebble Count % Fines	0.221 -0.491*D_bf	0.001	0.56	0.001	-
Change in Slow Habitat %	-0.507+ 0.893*Div	<0.001	0.56	<0.001	-

Note: all dependent variables expressed as (% below diversion) - (% above diversion)

2.6.4 Fine Sediment Metrics

Differences in volumetric and areal percent fines varied in magnitude, but tended to follow consistent patterns at individual study sites. Sites with more or less fine sediment in the downstream reach as compared to the upstream reference reach had fairly consistent positive or negative differences (respectively) across the five local fine sediment measurements (Figure 2.2).

The areal percent fines measure yielded a greater percentage of fines than the volumetric measurement in the majority of study reaches. In fast flowing zones, the areal measure yielded a greater amount of fines than the volumetric measurement in 28 of 40 study reaches, with an average measurement of 16.2% more fines. Slow zones were estimated to have a greater amount of fines via areal measurement in 38 of 40 study reaches, with an average of 53% more fines than the volumetric measurement. Interestingly, both methods measured a similar level of variability, with fast/slow coefficients of variations (CV) of 1.34/0.82 for the areal measurements as compared to 1.04/0.81 for the volumetric measure. Coefficients of variation were higher for fast flowing zones than slow flowing zones for both areal and volumetric measures (Appendix B).



Note: _1 = summer visits and _2 = fall visits

Figure 2.2 – Difference in percent fines between downstream diverted reach and upstream reference reach

The central hypothesis of this study was that fine sediments would increase downstream of diversion dams was confirmed, but the actual percentage of downstream reaches having a greater amount of fine sediment varied depending on measurement technique. Eighty percent of streams contained more fine sediment downstream when measured volumetrically in fast flowing zones as compared to 50% of streams having more downstream sediment when measured volumetrically in slow flowing zones (Appendix B). Areal measures detected more fine sediment below diversion dams in both fast and slow habitat at nearly 70% of study sites. Wilcoxon signed rank test showed that fast flowing zones had significantly higher average velocities ($p = 0.001$) than slow flowing zones, yet slow flowing zones had greater depths 60% of the time (non-significant, $p = 0.50$). Slow flowing zones also contained significantly more volumetric ($p < 0.001$) and surficial ($p < 0.001$) fine sediment.

2.6.5 Sediment Supply

As a surrogate for direct measures of sediment supply, several local and basin conditions were analyzed. Study site stream banks and beds were stable, with little evidence of localized mass wasting and minimal suspended load during low flow conditions. The collective geology of the study site drainage basins is a mix of Precambrian and Quaternary igneous and metamorphic material, with some glacial deposits. Cross comparison of dominant basin rock types and geologic history did not reveal specific basins with a substantially greater potential for fine sediment production than others.

2.7 Discussion

A combination of three key factors provides a weight of evidence of increased fine sediment downstream of diversion dams: (1) a greater amount of fine sediment using both volumetric and areal measures within fast flowing habitat, (2) more fines in slow habitat as compared to fast, and (3) a shift toward additional slow flowing habitat in diverted streams,. In contrast to earlier studies that struggled to discern significant changes in both channel form and sediment response related to flow diversion (Bohn & King 2000, Ryan 1997, Wesche *et al.* 1988), this study detected several significant shifts in fine sediment, channel form, and hydraulics. In support of the first hypothesis (that selected study channels have similar character above and below the diversion structure) Wilcoxon signed rank test results demonstrate that key characteristics of geomorphic context, as evidenced by slope and grain size, remain consistent. Second, I hypothesized that there were significant differences between various physical and hydraulic characteristics between the upstream reference reach and the downstream diverted reach. This was supported by significant differences in 33 of 43 measured variables and calculated metrics (Table 2.3). Notably, significant differences in fine sediment based on surficial measurements in both fast and slow zones, volumetric measurements of the fast flowing zones, and reach-wide pebble counts demonstrate that fine sediment effects are found in fast and slow zones using multiple measurement techniques. This is in contrast to results from a previous diversion study of similar stream types (Bohn & King 2000), where no significant difference was detected in the percent substrate less than 2 mm based on a 100 particle pebble count. Furthermore, Ryan (1997) was unable to detect morphological changes (as indicated by bankfull width) in channels other than wider pool-riffle streams, yet I detected differences between the upstream and downstream channel bankfull width across all sites ($p < 0.001$), possibly due to a larger sample size or more stringent site selection criteria. The failure to detect any difference in volumetric fines deposited in slowing flowing habitats in this study could have been due to difficulties recovering representative portions of fines from the cylinder sample.

Statistical models indicating significant associations between increased fine sediment below diversions and decreasing basin area, bankfull depth, and d_{84} (Table 2.4) suggest that small streams appear to be more susceptible to fine sediment accumulation than larger streams. Although fine sediments are readily winnowed by energetic flows (Lisle & Hilton 1999), the types of diversion structures encountered in this study often reduce peak flows up to 45% (Ryan 1997). The susceptibility of small streams to fine sediment accumulation suggests that removal of equivalent proportions of water from small and large streams may have a comparatively larger effect in smaller channels due to a higher surface area to volume ratio and the resulting greater influence of boundary and bank roughness.

The inverse relationship between d_{84} and fine sediment accumulation could indicate that streams with bed material loads composed of smaller particles are predisposed to flow diversion effects. Buffington & Montgomery (1999) argued that mountain streams receiving relatively high supplies of fine sediment should have higher bankfull dimensionless shear stress values. This study was unable to directly measure sediment supply; but I did examine relationships between bankfull dimensionless shear stresses and the significant variables of basin size and bankfull depth. A significant inverse relationship exists between bankfull dimensionless shear stress (referenced to d_{84})

and basin size was found ($p = 0.016$, Adjusted $R^2 = 0.24$), but no significant relationship to bankfull depth. Although I used the total bankfull shear stress as opposed to partitioning the grain shear fraction for the channel bed (Buffington & Montgomery 1999a), the inverse relationship suggests that the streams draining smaller basins in this study may receive higher loading of finer distributions of sediment, yet there is no association between basin size and the prevalence of "soft" rock types or glacial material.. The hydrographs of small snowmelt-dominated basins would likely have steeper falling limbs than larger basins, resulting in more well-sorted surface sediments, including fines (Hassan *et al.* 2006). The change in percent fines and d_{84} each quantify aspects of the sediment distribution, yet d_{84} is principally independent from the change in percent fines for two reasons: (1) the dependent variable, change in percent fine sediment, is normalized to the percent sediment in the upstream reference reach, whereas the independent variable d_{84} is a direct measure of the sediment in the upstream reach, and (2) when d_{84} is recalculated after truncating fines from the sediment distribution it is still significant ($p < 0.052$) in predicting change in fine sediment. The influence of each diversion's capacity to pass fine sediment and high flows could not be assessed due to the loss of too many degrees of freedom in the regression models of an already limited sample set. As large permanent diversion structures would be more effective than smaller temporary structures at capturing the peaks flow events, it is possible that these large structures would also capture and divert more fine sediments, though no evidence of this correlation was found at the flows observed during this study.

Evidence was found to support the first assertion of my fourth hypothesis that: (1) lower-gradient channels, (2) hydraulically rough channels, (3) channels with blockages, or (4) basins with a higher sediment supply are more susceptible to fine sediment deposition. Slope of the upstream reference reach was not found significant as a continuous variable predicting fine sediment deposition, yet differences in fine sediment response between sites greater and less than 3% slope possibly signal a threshold response of greater susceptibility of low-gradient channels to fine sediment accumulation. This result is consistent with the findings of Wesche *et al.* (1988), but a lack of sites less than 1.5% slope in this project precluded direct comparison at the same threshold.

With respect to the later assertions of the fourth hypothesis, no significant models were found relating channel Darcy's friction factor or ratio of wetted channel width to d_{84} (as an indicator of blockage potential) to any of the fine sediment measures. Although not statistically testable, there was no evidence that certain geologic settings are associated with a higher probability of fine sediment accumulation below diversions. This lack of evidence could be due to the overall small data set, the pristine nature of the study basins, or the distribution of slopes and roughness characteristics of the study sites. In general, the regression models developed in this study provide insight into factors that may predispose streams to fine sediment accumulation due to flow extraction, but more robust and transferable predictive models are precluded by a limited data set of 20 site visits, as well as the lack of a random site selection process.

2.7.1 Ecological Implications

A strong positive relationship between percent diversion and slow habitat ($p < 0.001$, Adjusted $R^2 = 0.56$) (Table 2.4) suggests that as more flow is extracted from streams, there is a greater shift from turbulent, well-mixed waters to tranquil conditions.

No significant differences were found in point measurements of water temperature or dissolved oxygen content between the upstream and downstream channels, most likely due to the proximity of the study reaches. However, a shift in macroinvertebrate composition away from rheophilic (or flow-loving) taxa was found at around 90% diverted (Albano 2006). This effect on habitat, when considered in combination with our finding of significantly less wetted bed area ($p = 0.004$), suggests that benthic habitat may be markedly altered below a diversion. These changes, in addition to the fore-mentioned increased fine sediment levels, could negatively influence macroinvertebrate and fish assemblages, contribute to colonization of in-channel macrophytes and periphyton, or allow riparian encroachment (Rader & Belish 1999, Waters 1995, Ligon *et al.* 1995).

2.7.2 Evaluation of Fine Sediment Measures

The development of multiple areal and volumetric fine sediment measurement schemes provided greater local-scale and reach-wide detail on the shifts in fine sediment than EPA's Environmental Monitoring and Assessment Program (EMAP) protocols which use a visual estimation of 110 particles (Faustini & Kaufmann 2007). Each of the five fine sediment metrics used in this study is characterized by inherent advantages and challenges in definition and application (Table 2.5).

For future studies focused on distributions of fine sediment, I would recommend the utility of the areal grid count, for its ability to (1) quickly assess a large area in a limited amount of time, (2) discern the impacts of a range of bed conditions from fines drape to bed armoring, and (3) applicability for any substrate size, including bedrock.

Table 2.5 – Advantages and challenges of fine sediment measures

Parameter	Advantages	Challenges
Volumetric % Fines – Fast	• Allows collective surface and subsurface analysis of fine sediments • If macroinvertebrate are being collected with a sampling cylinder, little additional work is required • Capable of measuring distribution of fines smaller than 2 mm	• Can be difficult to insert cylinder sampler in coarse substrate • Bemoulli effect of flowing water around cylinder has the ability to pull fluid and suspended fine sediments from inside of cylinder • Covers minimal area for required effort
Volumetric % Fines – Slow	• Cylinder sampler tends to seal with bed better in finer substrate • If macroinvertebrate are being collected with a sampling cylinder, little additional work is required • Not subject to skewing due to fines drape • Capable of measuring distribution of fines smaller than 2 mm	• Can be difficult to scoop specific depth of sediments out of cylinder without winnowing • If a large amount of fine sediments are available that will suspend, the fluid within the cylinder needs to be sampled as well • Covers minimal area for required effort
Areal % Fines – Fast **Areal % Fines – Slow**	• Quick visual analysis allowing for a large number of points • Adjustable spatial density sampling depending on sampling grid • Would work on any size or type of sediment, including bedrock • detects influence of fines drape or bed armor	• Quasi-subjective criteria on whether a point is 'fine' or not • Can be confounded by a drape of fines or vegetative matter on the bed • In this study, spatially restricted to areas around cylinder sites • Not capable of determining size distribution of sediment of any sediment only measures presence /absence of sediment less than 2 mm
Pebble Count % Fines	• Best spatially distributed sample over the entire reach • Allows full sediment size distribution for all clasts less than 2 mm	• Does not allow for size distribution of fine sediments less than 2 mm • Time consuming and best suited for gravel or cobble bed streams • Accuracy subject to total number of pebbles measured and measurement technique

2.7.3 Implications for Diversion Operation and Design

Diversion dams are a dominant component of the water distribution infrastructure, yet their operation and design can influence the effect on the downstream ecosystems. In many highly diverted systems, the establishment of environmental flows (often minimum) is primarily focused on the larger streams in the system. As demonstrated in this study sample of low-head diversion structures, fine sediment appears to accumulate even when peak flows are passed by the diversions, indicating that multiple elements of the undepleted flow regime may be necessary to sustain important geomorphic and ecological processes (Poff *et al.* 1997). In accordance with Arthington *et al.* (2006), additional effort is needed to evaluate and establish environmental flows on relatively small biologically-rich streams, including passage of moderate and high level flushing flows to mitigate fine sediment deposition, especially on streams of milder gradient. Beyond flow management, diversion dam design and construction could also influence the effects on the downstream channel. Structures designed to be submerged or with adjustable gates that would pass a portion of the naturally occurring base and peak flows would allow a more natural hydrograph than backwater structures that homogenize flow by only passing a set volume of water.

2.8 Conclusions

Natural streams are comprised of a heterogeneous mosaic of habitat, created by gradients of hydraulic variation and complex sedimentation patterns. In this study, flow extraction by diversion dams shifted this complex habitat assemblage towards lower flow velocities and more fine sediment. Of the 20 paired observations on 26 reaches, with flow diversion ranging from 23% to 99%, the channels most susceptible to fine sediment degradation are characterized as shallow, in small basins, and with smaller sized coarse substrate. The combination of a: (1) greater prevalence of fine sediment in slow versus fast habitat, (2) a shift toward more slow flowing zones, and (3) increased levels of fine sedimentation using four of five measures consistently indicates increased fine sediment deposition below diversion dams, particularly for channels of less than 3% slope. Findings based on patch-scale areal and surficial measurements are supported by the increased level of fine sediments within reach-wide pebble counts. In contrast to previous studies with generally inconclusive results regarding downstream effects of diversion dams, the strength of this study stems from careful selection of paired sites with consistent geomorphic setting, the application of multiple measurements of fine sediment, and the implementation of a detailed field protocol.

In future studies, streamflow gauging would provide valuable insights into levels of flow extraction over time, the influence of hydrograph characteristics, and episodic flow variation. A laboratory or fully controlled field study allowing controlled manipulation of diversions could provide greater insight into the direct relationships between magnitude of diversion and channel characteristics. Additionally, study of the ability of diversion structures to pass both channel maintenance flows and fine sediment could influence diversion design. The methods used for this project provide as a baseline for future studies of its type, yet further investigation of sediment sampling techniques is merited to help consider the strengths, limitations, and applicability of each.

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2.10 Symbols, Units, and Abbreviations

Symbols

Adjusted R^2	=	adjusted coefficient of determination
Basin	=	basin size in square kilometers
Cp	=	Mallow's ranking
CV	=	coefficients of variations
d_{16}	=	sediment diameter where 16% of total is smaller by size
d_{50}	=	average sediment size
d_{84}	=	sediment diameter where 84% of total is smaller by size
D_bf	=	bankfull depth
Div	=	percent diverted
n	=	sample size
p	=	p-value
PCFines	=	pebble count % fines
R^2	=	coefficient of determination
Re	=	Reynolds number
Slow	=	proportion slow
SurfFinesFast	=	areal % fines – fast
SurfFinesSlow	=	areal % fines – slow
VolFinesFast	=	volumetric % fines – fast
VolFinesSlow	=	volumetric % fines – slow

Units of Measure

cm	centimeter(s)
km^2	kilometer(s)
L/s	liter(s) per second
m	meter(s)
m/m	meter per meter
m/s	meter(s) per second
m^2	square meter(s)
m^2/s	square meter(s) per second
m^3/s	cubic meter(s) per second
mm	millimeter(s)
N/m^2	Newton(s) per square meter
%	percent
W/m	Watt(s) per meter
W/m^2	Watt(s) per square meter

Abbreviations

CDSS	Colorado's Decision Support Systems
ESRI	Environmental Systems Research Institute, Inc.
NID	National Inventory of Dams
PCA	principal components analysis

®	registered
STAR	Science to Achieve Results Program (USEPA)
™	Trademark
USA	United States of America
USACE	U.S. Army Corps of Engineers
USEPA	U. S. Environmental Protection Agency

CHAPTER 3

NUTRIENT UPTAKE AND TRANSIENT STORAGE OVER A GRADIENT OF GEOMORPHIC COMPLEXITY

3.1 Abstract

The understanding of nutrient uptake in streams is restricted by the limited consideration of the interactions of geomorphic setting and flow regime with biogeochemical processing. This project investigates physical and hydraulic components that influence transient storage and nutrient uptake in small agricultural and urban streams across a gradient of channel conditions and management modifications. Three geomorphically distinct segments on each of two streams were studied in the summer of 2007: one in a Colorado Front Range urban setting and the other in a mountainous agricultural region in north-central Colorado. The urban stream exhibits various levels of stabilization and planform alteration, and the agricultural stream has been subject to historically variable cattle-grazing practices. Reach-scale geomorphic complexity was characterized using highly detailed surveys of channel morphology, substrate, hydraulics, and habitat units. Injections of conservative bromide (Br^-) and non-conservative nitrate (NO_3^-) tracers were used to characterize channel processes. Geomorphic characteristics, specifically increased longitudinal roughness and flow depth, were strongly associated with both nutrient uptake and transient storage, thus underscoring the importance of geomorphic influences on stream biogeochemical processes.

3.2 Introduction

Land use changes and altered hydrologic and sediment regimes have resulted in widespread degradation and homogenization of physical habitat in urban and agricultural streams (Jacobson *et al.* 2001, Allan 2004). Urban streams are not only affected by an increase in volume and intensity of runoff, but also increased loading of nutrients, metals, pesticides, and other contaminants (Paul & Meyer 2001). In the United States, agriculture is associated with degradation of over 270,000 km of streams and rivers, closely followed by hydromodification and urban impacts (USGS 2000).

Land use alterations often destabilize channels and degrade physical habitat to an extent that prompts stream rehabilitation activities. As opposed to restoring the dynamic processes that create a diverse habitat template, most rehabilitation projects are focused on building static control features (Wohl *et al.* 2005, Bernhardt *et al.* 2005). Concurrent with the burgeoning interest in reversing the effects of land use change on streams, there has been a growing scientific interest in the nutrient uptake functions of small streams and their influence on downstream water quality, particularly nitrogen (N) enrichment (Peterson *et al.* 2001, Alexander *et al.* 2007, Mulholland *et al.* 2008b, Craig *et al.* 2008) However, there is little known about the connection between nutrient uptake and the habitat templates rehabilitation activities endeavor to create.

This study is aimed at quantifying the geomorphic complexity of agricultural and urban streams across a gradient of human modifications and rehabilitation to examining its influence on transient storage and nitrate uptake. The term "geomorphic complexity" is used to refer to the multi-scale assemblage of physical components that make up a stream channel, ranging from the spatial distribution of patch-scale substrate and hydraulic characteristics to the reach-scale topographic variability of channel cross-sectional, longitudinal, and planform profiles. This complexity in streams and rivers has been linked to a variety of physical, biological, and biogeochemical processes, including fish and benthic macroinvertebrate habitat, organic matter retention and processing, transient storage, hyporheic exchange, and fluxes of energy and materials (Allan 2004, Sheldon & Thoms 2006, Buffington & Tonina 2009). In the present study, I developed a procedure for quantifying various forms of geomorphic complexity and related these forms to the transient storage capacity of the study streams. I subsequently examined the combined influence of geomorphic complexity, flow hydraulics, transient storage, and other contributing factors on nitrate uptake.

3.3 Background

Nitrogen is an essential nutrient in both terrestrial and aquatic ecosystems; it regulates the level of primary production and is an essential building block of all proteins (Manahan 2004). Excess nitrogen has been linked to many forms of environmental degradation, including coastal eutrophication (Rabalais *et al.* 2002), release of greenhouse gases (Reilly *et al.* 2002), and saturation of forest ecosystems (Fenn *et al.* 1998). Through agricultural use of nitrogen fertilizers, humans have roughly doubled the availability of fixed nitrogen on our planet, with ongoing increases forecast for the foreseeable future (Tilman *et al.* 2001). This augmented flux of nitrogen is ultimately drained into our rivers, estuaries and coastal waters, leading to eutrophication and associated hypoxia (Carpenter *et al.* 1998, Green *et al.* 2004).

River networks can be a critical link in the transport and fate of nutrients as they provide an important interface between terrestrial and aquatic systems and offer a dynamic and highly heterogeneous environment for optimizing biogeochemical processing (McClain *et al.* 2003). As water flows downstream, it continues interacting with the landscape, exchanging with groundwater and biotic assemblages residing in the channels. Thus, dissolved nutrients can be assimilated into plant material, adsorbed into particulate matter, released back to the water column, or given appropriate conditions, permanently removed from the ecosystem. The theory of nutrient spiraling was developed to describe the simultaneous downstream transport and cyclical biotic processing of these nutrients in advective aquatic systems (Newbold *et al.* 1981, Webster & Patten 1979). While algal uptake is possible in the water column, it is believed that the majority of biotic processing takes place on and within the bed of the channel (Boulton *et al.* 1998). This processing can be measured by several methods that range from enzyme assays of sediment slurries (Strauss & Lamberti 2000), providing a measurement of potential reaction rates, to tracer injections into natural streams (Gooseff *et al.* 2007, Peterson *et al.* 2001, Tank *et al.* 2000, O'Connor & Harvey 2008, Böhlke *et al.* 2004, Harvey & Wagner 2000). While the assays provide the greatest control yet offer minimal replication of natural conditions, the tracer injections realize the greatest realism but provide the least control. Intermediate on the scale is the study of sediment perfusion cores (Sheibley *et al.* 2003) performed in a laboratory, but using intact sediment cores with groundwater percolated through them. Recent work using the USGS MINIPOINT sampler has allowed in-situ, small scale measurements of nutrient uptake in the pore water space of the hyporheic zone (Böhlke *et al.*2009).

Small streams, typically comprise two-thirds to three-quarters of drainage network length (Leopold *et al.* 1964) and have large surface-to-volume ratios that favor rapid uptake and processing of nutrients (Peterson *et al.* 2001). During seasons of high biological activity, headwater streams typically export downstream less than half the input of dissolved organic nitrogen from their watersheds (Peterson *et al.* 2001). Additionally, in a 12 year weekly study of a first-order forested stream in Tennessee, Mulholland (2004) reported stream processes removing 20% of the NO_3^- entering the stream annually. Network models suggest that small streams maintain a higher biotic removal efficiency of NO_3^- than larger streams for all levels of NO_3^- flux (Alexander *et al.* 2000). Yet, at high levels of NO_3^- loading, the removal capacities of both small and large streams are critically overloaded, leading to unchecked rates of nutrient export (Mulholland *et al.* 2008b). According to a review of contemporary literature, the greatest opportunity for nitrogen removal exists in small streams carrying large loads of nutrients at low to moderate flows (Craig *et al.* 2008). Well functioning stream networks can, therefore, regulate the export of nutrients from the landscape and ameliorate the detrimental effects of eutrophication in downstream ecosystems (Alexander *et al.* 2007, Alexander *et al.* 2000).

Owing to its prevalence, reactivity, and mobility, nitrogen serves as a surrogate for other contaminants in water where the downstream transport and effects are of concern. Inorganic nitrogen is available in two main forms in aquatic systems, as ammonium (NH_4^+) and as nitrate (NO_3^-). Ammonium tends to be less prevalent in the water column, due to its affinity to immobilization, often adsorbing onto clay particles/organic matter, or nitrification into NO_3^-. Nitrate is highly soluble and easily leached from agricultural and urban landscapes into adjacent waterways. Nitrogen can be transformed by processes including assimilation into biofilms and vegetative matter, adsorption into stream sediments, immobilization by microbes degrading organic substrates, transformation among several molecular states, or when conditions are optimal, transformation and release as N_2 gas (Manahan 2004). This transformation into N_2 gas, known as denitrification, is the result of a series of bacterial conversions from NO_3^- into N_2, and is the primary means of permanent nitrogen removal from aquatic ecosystems. Most other processes are transient, with the N moving further downstream in plant material or released to the water column through decomposition or other processes. Denitrification by streambed bacteria requires the optimal conditions of minimal oxygen content, available nitrate, and ample organic carbon as an energy source. Various processes support these requirements; modeled stream storage zones have lower oxygen concentrations than the main channel (Chapra & Runkel 1999) and organic debris dams and organic-rich gravel bars in urban streams enhance denitrification potential (Groffman *et al.* 2005). Denitrification has been shown to be a substantial permanent sink for N in an agricultural stream at baseflow conditions (Böhlke *et al.* 2004), yet too much nitrogen can lead to N saturation, which contributes to non-linear decreases in nitrate uptake efficiency (Earl *et al.* 2006). Recently, compartmental analysis, often used in groundwater modeling, has shown promise for modeling nutrient assimilation (Faulkner & Campana 2007).

The use of isotopically-labeled nutrients, such as ^{15}N-ammonium (Peterson *et al.* 1997, Hall *et al.* 1998, Peterson *et al.* 2001, Webster *et al.* 2003) and ^{15}N-nitrate (Hall *et al.* In-prep, Mulholland *et al.* 2008b, Böhlke *et al.* 2004) has helped to better quantify the overall fate of nitrogen within a stream system. Although costly, these labeled tracer studies are increasingly used to understand and compartmentalize nitrogen fate and transport. One highly influential stable isotopic tracer study is the Lotic Intersite Nitrogen Experiment (LINX), which has been carried out in two phases: LINX I (1996 to 2001), focused on ammonium NH_4^+ uptake in minimally disturbed headwater streams, and LINX II (2001 to 2006) focused on NO_3^- uptake across a gradient of land use. LINX I contrasted NH_4^+ short-term enrichment plateaus with a single longer-term $^{15}NO_3^-$ tracer addition. Results suggested that uptake lengths (S_w) derived from the short-term enrichment plateaus were significantly longer than those derived from the isotopic tracer injection. Additionally, LINX I concluded that the ratio of these two uptake lengths was proportional to the level of NH_4^+ in the reach, meaning that nutrient addition experiments could still be used as long as the level of addition was kept to a minimum (Mulholland *et al.* 2002). LINX II subsequently investigated nine streams in each of eight geographically diverse regions, analyzing three agricultural, urban, and reference streams in each region. Use of ^{15}N-nitrate minimized elevation of ambient nitrate concentration to ca. 7.5%, thereby minimizing effects on uptake rates. LINX II found measurable NO_3^- uptake in 69 of 72 streams and denitrification in 49 of 72 streams, yet denitrification only accounted for an average of 16% of total uptake. Using structural equation modeling, LINX II found weak relationships between denitrification and transient storage parameters, a decline in uptake efficiency with higher levels of NO_3^- concentration, and a negative relationship between uptake length and gross primary productivity (GPP). They found no significant difference in uptake between land use categories (Hall *et al.* In-prep, Mulholland *et al.* 2008b). In contrast to the LINX II findings, Tank *et al.* (2008) found no significant decline in uptake velocity with increasing NO_3^- loading among 227 previous uptake measurements. LINX II found that small streams (< 100 L/s) had the greatest influence at low levels of NO_3^- loading, larger streams (≥ 100 L/s) had increasing importance at moderate loadings of NO_3^-, but all streams were subject to diminished uptake efficiency at high levels of NO_3^- (Mulholland *et al.* 2008a).

As the majority nutrient loading is now derived from human activities, streams in agricultural and urban areas often carry elevated nutrient loads (Walsh *et al.* 2005). Altered streamflow and sediment regimes in these agricultural and urban streams often lead to decreases in geomorphic complexity via channel straightening, enlargement, or sediment aggradation, removal of woody debris and bank vegetation, and armoring of naturally variable banks (Allan 2004). The complexity of a stream segment is a function of the water and sediment inputs to the system, interacting with the parent lithology, antecedent channel form, and continual or punctuated disturbances to the system. Naturally variable fluxes of these inputs tend to lead to a dynamic assemblage of hydraulic and geomorphic features. The scale that at which this patchiness occurs is highly variable in time and space (Palmer & Poff 1997, Jacobson *et al.* 2001).

To improve the quality of many waterways, a combined strategy of watershed best management practices and channel restoration has been implemented. Over $1 billion is being spent on stream restoration activities annually in the United States (Palmer *et al.* 2007). While this level of restoration has the potential to substantially enhance ecological stream functions (including nutrient processing), channels are often restored for the individual improvements of fish habitat, aesthetics, or bank stability. Standards have been developed to help ensure ecologically sound restoration strategies

(Kondolf 1995, Palmer *et al.* 2005), yet even when ecological concerns do motivate stream restoration projects, post-project aesthetics and a generally positive level of public satisfaction are the most commonly used metrics of success (Bernhardt *et al.* 2007). The ecological implications of increasing channel complexity through both active and passive rehabilitation are poorly understood, especially with respect to nutrient uptake. Furthermore, there is a paucity of studies examining the effects of stream rehabilitation on nitrate spiraling, thus the efficacy of stream rehabilitation as a means of reducing downstream nitrate loading is uncertain. In one notable exception, restoration of a channelized stream through construction of pool-riffle sequences and reconnection to the former floodplain reduced both flow velocity and downstream transport of nitrogen (Bukaveckas 2007).

Scientific understanding of nutrient processing, fate, and uptake rates has steadily progressed, while the linkage to the geomorphic setting has been slower to develop. Geomorphic complexity is a key influence to creating areas of storage within a channel. These physical compartments of temporary water retention, known collectively as transient storage, provide nutrient-rich stream water greater exposure to substrates, biota, and specific conditions for nutrient processing. To assist the reader's understanding of these complex causal pathways, the following sections sequentially discuss geomorphic complexity, transient storage, and nutrient uptake and then briefly summarize the linkages among them.

3.3.1 Influence and Measures of Geomorphic Complexity

Heterogeneity in streams is a vital component of lotic processes; however, variability in physical factors is easier to show than the evidence of biological importance of that variability (Palmer & Poff 1997). Streams are composed of a mosaic of habitat patches, each consisting of a unique combination of textural, geomorphic, hydraulic, and biological traits that collectively determines the function of that patch (Pringle *et al.* 1988, Lancaster 2000, Brooks *et al.* 2005). Individual combinations of sediment texture can be stratified into detailed textural facies (Buffington & Montgomery 1999b) and meso-scale topographical features are known to create highly complex flow patterns (Crowder & Diplas 2006). Additionally, a variable and irregular downstream geometry reduces the development of clogging layers of fine sediment (Schalchli 1992). Bartley and Rutherfurd (2005) performed an exhaustive evaluation of thalweg, cross-sectional, and sediment size variability measures in order to determine which specific measures were most applicable at particular spatial scales. Sheldon and Thoms (2006) used multiple metrics for cross-sectional size, shape, and complexity as related to organic matter retention. Of particular relevance in this study are the dead zones, blockages, backwater effects, hyporheic exchange, and complex hydraulics generated by geomorphic complexity, and their influence on transient storage.

3.3.2 Functional Significance of Transient Storage

Transient storage is defined as any physical aspect of a stream temporarily retaining water, and it includes both in-channel areas (pools, eddies, and channel margins) and the margin around the streambed, known as the hyporheic zone. In the hyporheic zone, stream water flows through interstitial spaces of stream sediments on variable temporal and spatial pathways. The spatial scale of hyporheic flow can vary from seepage around a single particle on the bed to kilometer-scale flow through point bars and wide floodplains (Stanford & Ward 1993). While the influence of hyporheic processes on nutrient uptake has been known for over two decades (Grimm & Fisher 1984, Triska *et al.* 1989, Valett *et al.* 1997), the development of causal understanding has taken longer to evolve.

Hyporheic exchange results from the interaction between the hydraulic pressure of the flowing stream, channel geomorphic complexity, and the hydraulic conductivity of the streambed and banks. The magnitude of vertical hydraulic pressure gradients has been shown to be positively correlated with increasing average water surface concavity (AWSC) and spacing between zones of flow upwelling from and downwelling to the hyporheic zone (Anderson *et al.* 2005). Data obtained from well networks and groundwater models can be used to examine hyporheic exchange in high-gradient step-pool sequences, as well as in low-gradient channels with complex lateral structures and sinuosity (Kasahara & Wondzell 2003). Bed sediments have been shown to be highly heterogeneous and non-isotropic, thus confounding standard simplifying assumptions about substrate homogeneity in many models (Cardenas & Zlotnik 2003). Bedforms, as represented by sinusoidal pressure head distributions, can be the primary drivers of hyporheic zone flux in three-dimensional (3-D) subsurface models (Cardenas *et al.* 2004). Gooseff *et al.* (2007) demonstrated that a metric of geomorphic complexity defined as the product of slope, sinuosity, and longitudinal roughness was associated with increased potential for transient storage over a gradient of land use. Geomorphic complexity influences in-channel storage through increased channel roughness, greater extent of backwater, and creation of more tortuous flow paths. Removal of coarse woody debris and vegetation has been shown to decrease transient storage whereas construction of flow baffles increases storage (Ensign & Doyle 2005).

Strong gradients of oxygen concentration and organic carbon availability, coupled with greater travel time for surface water passing through the microbially-rich hyporheic zone, create the potential for high metabolic activity and can significantly influence nutrient dynamics (Mulholland & DeAngelis 2000). Concurrent with the development of a transient storage model (TSM) (Bencala & Walters 1983), the relationship between surface/groundwater exchange and biogeochemical processing was established (Grimm & Fisher 1984, Newbold *et al.* 1981). Nutrient uptake lengths are inversely related to the prevalence of transient storage zones, which differ among streams as a function of geomorphic attributes (Valett *et al.* 1996). Peterson *et al.* (2001) found that the hydraulic influences of flow depth and current velocity were directly related to NH_4^+ uptake lengths. Transient storage and discharge were closely related to soluble reactive phosphorus uptake lengths, while substantial differences in land use were not in three agricultural watersheds in Oklahoma (Haggard *et al.* 2001). As further evidence of its biogeochemical importance, the hyporheic zone has also been shown to provide the majority of whole system metabolic respiration within a stream (Naegeli & Uehlinger 1997).

3.3.3 Transient Storage Modeling

Documentation from the Stream Solute Workshop (1990) provides a detailed overview of the one-dimensional (1-D) TSM which integrates physical, chemical, and biological processes that mediate the interaction between hyporheic exchange and the fate of stream nutrients. With the development of the OTIS (One-dimensional Transport with Inflow and Storage) model by the U. S. Geological Survey (USGS), the TSM allows users to fit transient storage parameters to breakthrough curve (BTC) data in an automated environment (Runkel 1998). The TSM is robust in its incorporation of advection, dispersion, lateral inflow, uptake, and assimilation, yet it is limited by a single, exponential residence time distribution and a non-advective storage zone (Gooseff *et al.* 2005). Other models have been developed to better represent the long tail of the residence time distribution (Haggerty *et al.* 2002, Gooseff *et al.* 2003) . Progress has also been made to quantify hyporheic exchange through an empirically derived effective diffusion coefficient, thereby circumventing the challenges of tracer modeling and analyzing smaller spatial scales (O'Connor & Harvey 2008). Using combined terms for all forms of streambed transport, a vertical dispersion model estimates solute transfer rates in streambeds with wavy boundary conditions (Qian *et al.* 2008).

Just as the understanding of transient storage has evolved, so have the parameters used to describe it. Initially, the ratio of storage area to channel area (A_S/A) was the principal metric used to estimate the fraction of transient storage, but this measure did not account for the exchange rate between the storage zone and main channel (Valett *et al.* 1996). Runkel (2002) formulated the term F_{med}^{200} (Equation (3.1)), the fraction of median travel time due to transient storage, to represent the interaction between downstream velocity (u), A_S, and exchange coefficient (α), normalized to a 200 m reach length. F_{med}^{200} has found favor for being dimensionless and factoring in stream velocity and exchange rate between the storage zones and main channel better than other transient storage metrics.

Equation (3.1): Empirical equation for F_{med}^{200} – fraction of median travel time due to transient storage

$$F_{med}^{200} \cong \left(1 - e^{-L\frac{\alpha}{u}}\right)\frac{A_S}{A + A_S}$$

The dimensionless Damkohler number (DaI) has been formulated as an index of the influence of transient storage on reach-scale advective transport (Wagner & Harvey 1997) (Equation (3.2)). Ideal Damkohler values for representative transient storage modeling are on the order of 1.0, as when DaI << 1, minimal solute enters the storage zone, but when DaI >> 1, the reach is too long to pick up the signature of the storage zone on the solute breakthrough curve.

Equation (3.2): Damkohler number

$$\mathrm{DaI} = \alpha\frac{(1 + A/A_S)L}{u}$$

Hyporheic exchange has been shown to decrease with increasing unit discharge (Hall *et al.* 2002) and increase in larger streams per area of streambed (Kasahara & Wondzell 2003). Solute injection and sampling strategy also have a significant effect on the reliability of parameter estimates. Harvey *et al.* (1996) found that the use of the stream tracer technique adequately quantifies exchange with shallow alluvium over the time scale of hours, but it is relatively insensitive to slower exchange with deeper sediments. Constant injections with sampling all through the rise, plateau, and fall have provided better results than a pulse injection (Wagner & Harvey 1997). While the biochemical importance of in-channel and hyporheic storage varies greatly, only a few studies have been able to separate the two (Gooseff *et al.* 2003, Salehin *et al.* 2003, Briggs *et al.* 2009).

3.3.4 Nutrient Uptake Modeling

Along with the development of transient storage modeling techniques, the combined use of conservative and non-conservative tracers in short-term injections has allowed the separation of transient storage from nutrient uptake processes. The initial metric for nutrient uptake, as originally specified by Newbold *et al.* (1981) and later outlined by the Stream Solute Workshop (1990), is uptake length (S_w), or the average distance traveled by dissolved nutrients in the water column before uptake. Uptake length is a measure of uptake efficiency (as it is dependent on supply) rather than a true rate. According to the uptake length methodology, constant-rate nutrient injections are elevated in a stair-step fashion, to produce multiple plateaus of increasing concentration. During the plateaus, downstream concentration profiles are collected and the decline in nutrient concentration is used to calculate a longitudinal uptake rate (Webster & Valett 2007). Even with improvements to the method (Payn *et al.* 2005) the method was still limited by its linkage to hydrologic processes (i.e., transient storage, inflow, and dispersion). Runkel (2007) proposed a transport-based approach that parses the hydrologic from non-hydrologic processes and estimates uptake and transient storage parameters independently. Single, short-term, non-plateau injections are measured at one or more downstream stations, limiting the amount of tracer released to the streams, thereby saving time and eliminating the channel disturbance of longitudinal sampling. BTCs are then fit to the OTIS model (Runkel 1998) and parameter estimation can either be performed by OTIS-P, the predictive component of the OTIS package, or inverse modeling packages such as UCODE (Poeter & Hill 1999, Scott *et al.* 2003). Other nutrient uptake metrics include areal uptake rate, U (a mass rate measure of biotic uptake) and uptake velocity (v_f analogous to the vertical velocity at which a solute moves through the sediment/water interface). The metric v_f is preferred for cross-site comparisons for its independence from stream velocity and nutrient concentration.

3.4 Linkage of Geomorphic Complexity, Transient Storage, and Nutrient Uptake

The impact of geomorphic complexity on nutrient uptake lies in the heterogeneity that the geomorphic template imposes on hydraulics, transient storage, and organic matter retention of a stream. Geomorphic features and localized habitat structure influence the patchiness of stream substrates and micro-scale flow dynamics. Strong relationships between storage zone size and ammonium retention potential were found on headwater streams in Brazil (Gucker & Boechat 2004). In particular, flow obstructions such as vegetation and coarse woody debris can increase in-channel storage and nutrient uptake (Ensign & Doyle 2005). For example, additions of coarse woody debris to a headwater channel also reduced channel velocity, increased transient storage, and improved ammonium uptake (Roberts *et al.* 2007). The most robust metric of transient storage, F_{med}^{200}, is negatively correlated with S_w in a review of contemporary nutrient spiraling literature (Ensign & Doyle 2006) and inversely related to NO_3^- denitrification S_w in the recent LINX II study (P. Mulholland, pers. comm. 2008). This suggests that improved description of geomorphic features, such as blockages and longitudinal variation, has the potential to improve our understanding of variability in uptake among geomorphic settings. Additionally, few studies have performed repeated tracer injections at the same site to examine the influence of flow level on transient storage (Wondzell 2006, D'Angelo *et al.* 1993) and nutrient uptake (Valett *et al.* 1997) with most favoring cross-stream comparisons.

3.5 Objectives

To examine the geomorphic and hydraulic characteristics that support transient storage and nutrient processing, I used detailed channel and physical habitat surveys coupled with conservative and non-conservative tracer injections at three sites along each of two streams. The objectives of the project are three-fold: (1) to explicitly define and measure multiple forms of geomorphic complexity, (2) to quantify the level of transient storage and nutrient uptake in six stream reaches across a gradient of geomorphic setting and land use, and (3) to determine the physical and hydraulic channel conditions that most directly influence hyporheic exchange and nutrient uptake, with the goal of providing sound information on the physical characteristics in small urban and agriculture streams during low flow conditions associated with increased nitrate uptake. To meet these objectives I proposed the following hypotheses:

1. reach-scale geomorphic complexity differs in form and magnitude among study sites;

2. increased levels of various forms of geomorphic complexity result in significantly more transient storage; and
3. increased levels of geomorphic complexity, transient storage, reach-wide metabolic activity, and/or benthic carbon significantly increase nitrate uptake.

3.6 Methods

3.6.1 Study Reach Selection

Three 180 m reaches on each of two streams were selected; one stream in a Colorado Front Range urban setting and the other in a mountainous agricultural region in north-central Colorado. Each segment was chosen for its distinctive geomorphic setting and historical human modification; the urban stream showing various levels of stabilization and planform alteration, and the agricultural stream subject to variable cattle grazing practices. All segments were surveyed using a detailed protocol for characterizing physical complexity in terms of habitat units with distinct combinations of geomorphic, substrate, and hydraulic attributes. Study reaches were chosen to conform to the following criteria: (1) no major reach-scale changes in slope or geology, (2) no notable sediment or flow additions, and (3) no dams, diversions, or similar flow altering structures.

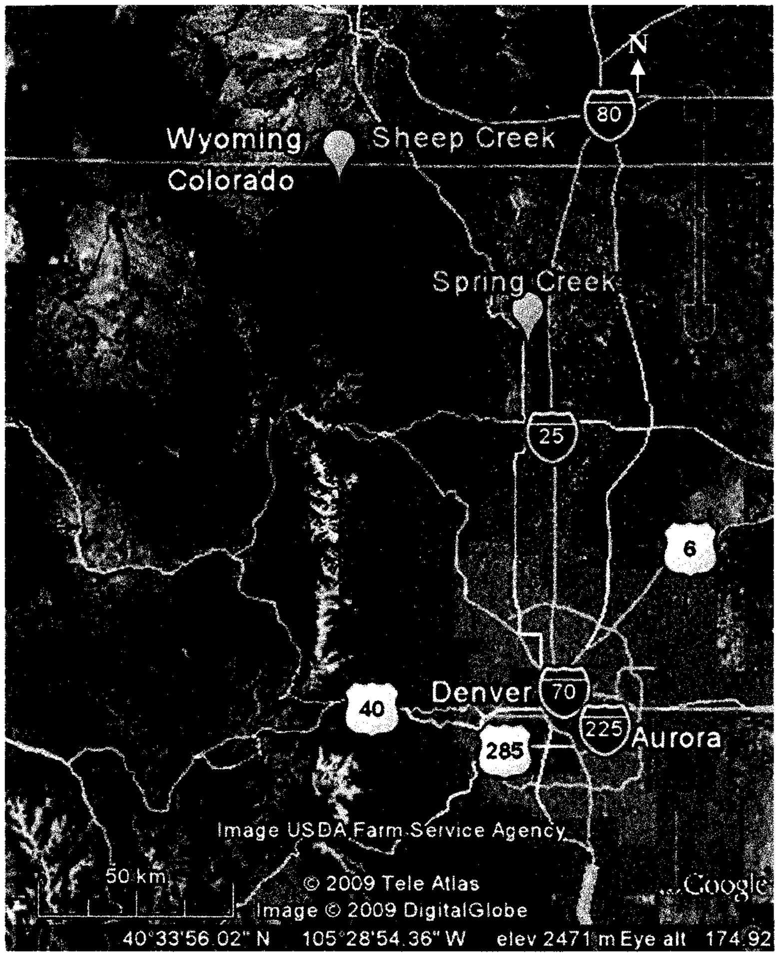


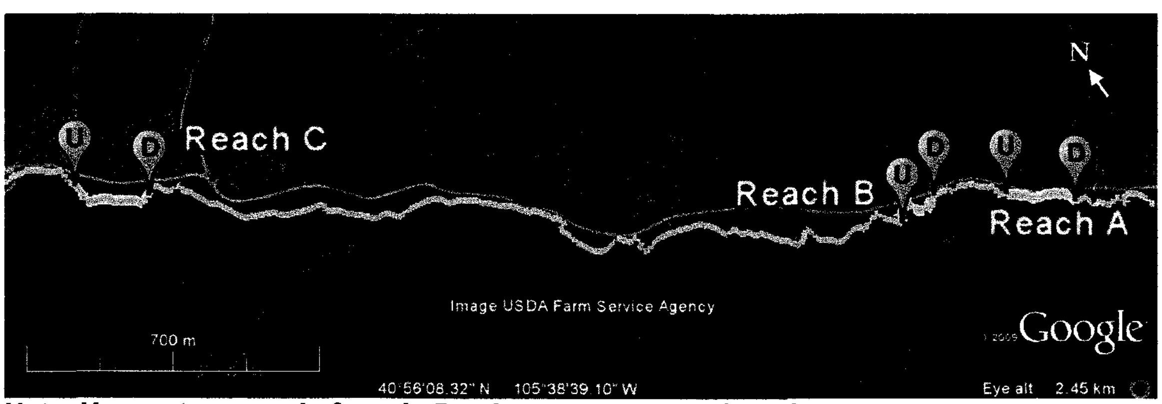
Figure 3.1 – Location of study sites

3.6.1.1 Agricultural Site – Sheep Creek

Sheep Creek is located 80 km northwest of Fort Collins, Colorado (N 40°55'48", W 105°38'16"), within the Roosevelt National Forest at an elevation of 2530 m. The flow regime is controlled by an upstream reservoir that is filled in the spring to early summer and drained via Sheep Creek in the late summer, causing an uncharacteristic dual peak in the hydrograph (the first due to snowmelt in the spring, and the next due to late summer water transfer, as seen later in Figure 3.16). Spring runoff is delayed until the reservoir is filled and some years the stream does not reach typical summer base flow until early fall. The National Forest Service land around Sheep Creek is used for cattle grazing in the summer months. The riparian area has been heavily grazed since the 1890s, but in 1956 the U.S. Forest Service fenced sections to exclude cattle use along 2.5 km of the stream for the improvement of trout habitat. These grazing exclosures have been related to shifts in vegetation patterns and nutrient dynamics that have been studied in detail by range scientists (Mergen *et al.* 2001, Flenniken *et al.* 2001, Stednick & Fernald 1999, Phillips *et al.* 1999). Each reach was selected to have a unique combination of grazing and morphological type. (Table 3.1 lists the grazing and morphological setting, Figure 3.2 is an overview map, and Figure 3.4 shows representative photographs of each reach.)

Table 3.1 – Summary of Sheep Creek reach settings

Reach	Character	Grazed	Bed Type	Sinuosity	Slope	d_{50} (mm)
Sheep A	Largely incised inside heavy brush and tree lined banks, coarse substrate.	No	Plane	1.08	1.5%	67.0
Sheep B	Highly sinuous with grass and brush lined banks, finer substrate.	No	Pool-riffle	1.91	0.7%	47.6
Sheep C	Grass lined banks, moderately incised, coarse substrate.	Yes	Plane	1.24	1.2%	54.1



Note: U = upstream end of reach, D = downstream end of reach

Figure 3.2 – Sheep Creek overview map

3.6.1.2 Urban Site – Spring Creek

The urban stream selected for this study was Spring Creek located in Fort Collins, Colorado, USA (N 40°30'50", W 105°4'7"), at an elevation of 1500 m. The upper portions of the watershed have been truncated by Horsetooth Reservoir and the existing watershed lies completely within Fort Collins. The use of the stream as a stormwater corridor and the flashy urban flow regime (spurring five major floods in the last 75 years), have led the city to straighten and stabilize significant portions of the streambed and banks for flood mitigation. High levels of public exposure to the stream via an adjacent hiking and biking trail, and proximity to residential landowners keeps the aesthetics of this stream a high priority. The three reaches were selected for their distribution of stabilization or alteration, with Edora reach being the most natural, Railroad reach having a history of modification but few structures, and Stuart reach having both bed and bank stabilization. (Table 3.2 lists the setting of each reach, Figure 3.3 is the overview map, and Figure 3.4 shows representative photographs of each reach)

Table 3.2 – Summary of Spring Creek reach settings

Reach	Character	Sinuosity	Slope	d_{50} (mm)
Edora	Set in a large city park, the most 'natural' of the three reaches.	1.16	0.4%	14.4
Stuart	Lateral and grade controlled with large grouted block bank protection with grouted boulder steps.	1.05	0.9%	34.7
Railroad	Straightened to allow grading of the adjacent property. Silty-clay bed with heavy grass lining the banks.	1.01	0.2%	< 2.0

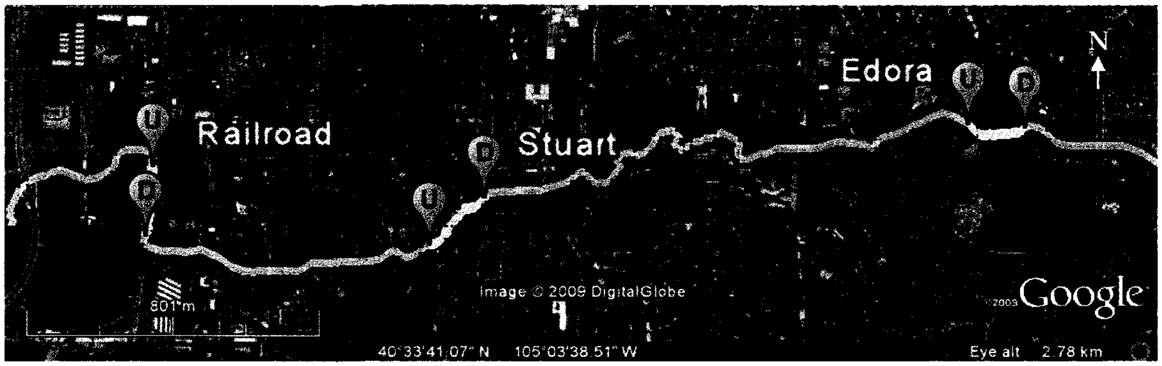


Figure 3.3 – Spring Creek overview map

3.6.1.3 Multiple Tracer Injections

The three study reaches on Spring Creek were selected for three separate nutrient injections at varying flows throughout the summer of 2007. The multiple injections were designed to analyze the influence of flow rate on transient storage and nutrient uptake. The initial injection was performed in mid June, a second in late July, and the third in mid-August; respectively, labeled X, Y, and Z (Table 3.3). An unexpected small flood passed through the corridor immediately after injection Y, scouring fines and organic matter from the substrate and laying down bank vegetation. The effects of this small flood will be examined in greater detail in later work by my colleague, Jenny Mueller Price (Mueller Price In-prep).

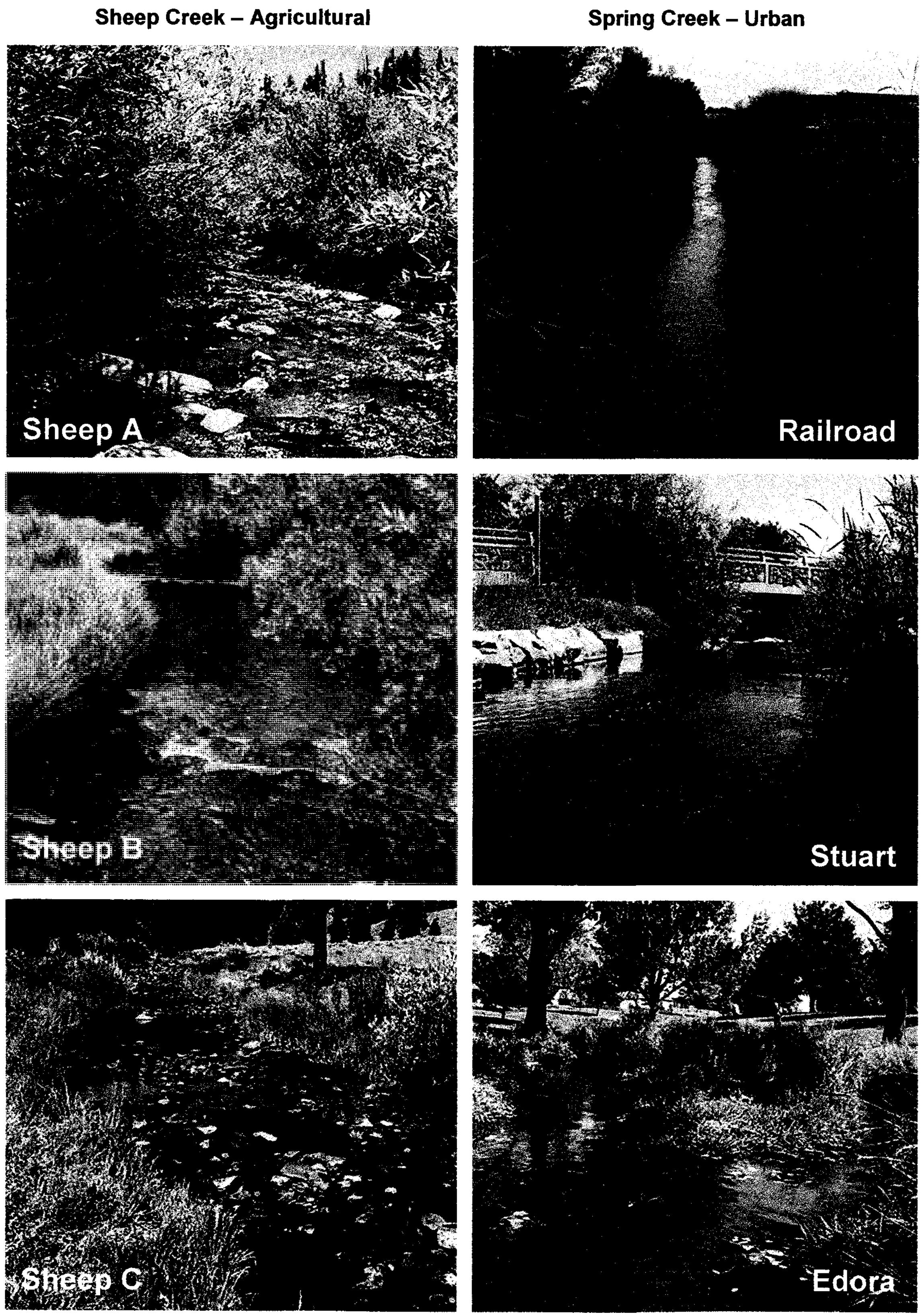


Figure 3.4 – Representative photographs of Sheep Creek and Spring Creek reaches

Table 3.3 – General study reach conditions and site/injection naming structure

Stream Reach	Site Code	Reach Length (m)	Injection Code	Date of Field Visit	Flow Rate, Q (L/s)[1]	Successful BTC Model
Sheep Creek A	ShA	184.1	ShA_X	7/15/2007	108.0	yes
Sheep Creek B	ShB	192.2	ShB_X	7/16/2007	88.0	no
Sheep Creek C	ShC	190.6	ShC_X	7/17/2007	102.0	yes
Spring Creek Edora	SpE	177.9	SpE_X	6/25/2007	156.5	no
			SpE_Y	7/30/2007	72.0	yes
			SpE_Z	8/6/2007	152.0	yes
Spring Creek Railroad	SpR	179.7	SpR_X	6/28/2007	70.0	no
			SpR_Y	8/1/2007	16.5	yes
			SpR_Z	8/8/2007	20.6	yes
Spring Creek Stuart	SpS	180.0	SpS_X	6/26/2007	133.0	no
			SpS_Y	7/31/2007	46.0	yes
			SpS_Z	8/9/2007	108.0	yes

[1] **Flow rates at all six sites were at low to moderate baseflow levels as compared to contemporary and historical stage records**

3.6.2 Stream Classification

Classification of stream character was performed using a top down spatial hierarchy, with stream type identified for each reach, followed by detailed geomorphic unit mapping and analysis First, the process-based stream typology of Montgomery and Buffington (1997) was used to identify the morphological character of each reach. Then, I quantified the patch-scale variation by dividing the stream into unique patches of geomorphic, hydraulic, and sediment character. These meso-scale patches (2 to 15 m^2 in area) spanned the width of the channel and often formed a repeating spatial pattern along the channel. Each stream was classified into one to eight different types of habitat units per reach. Habitat units were delineated by bed area, yet only habitat units intersected by designated transects were measured to define the physical and hydraulic characteristics of the unit. If any habitat unit type did not include a transect, an additional transect was added. All patch delineation and identification was performed by a single individual (myself) to eliminate inter-observer variation.

3.6.3 Field Measurements

Each study reach was divided by 21 equally spaced transects, 9 m apart, establishing a reference grid for habitat units, the channel survey, and other measurements along the reach. Next, the cross-sectional geometry of each transect and the downstream profile of the reach were surveyed with a total station, providing detailed planform and cross-sectional information. Cross-sectional points were recorded at least every 0.5 m across the channel and at every significant break in cross-sectional slope. Both the thalweg profile and channel width were surveyed every 3 m downstream, or at breaks in curvature or slope. This survey was the basis for the majority of geomorphic complexity metrics used in the project (Figure 3.5).

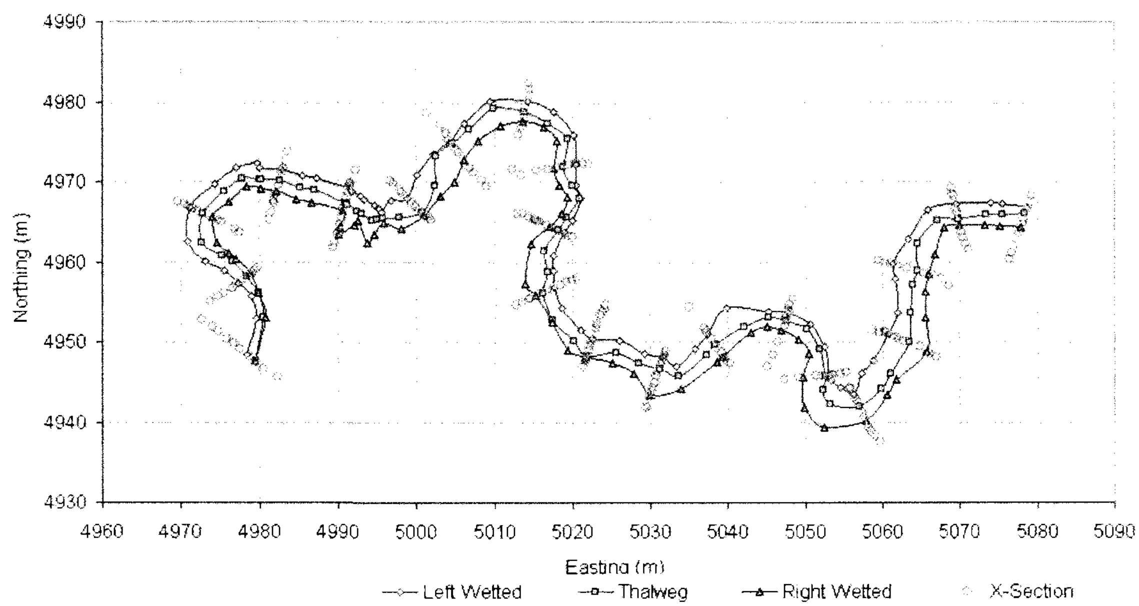


Figure 3.5 – Example planview survey – Sheep B reach, showing the survey detail of each cross-section as well as intermediate measures of channel width and thalweg location and depth

Depth-averaged velocity and flow depth were measured at five points across each cross-section with a Marsh McBirney Flo-Mate™ electromagnetic velocimeter (Hach Flow, Loveland, Colorado, USA) on a calibrated wading rod. The flow rate was quantified by 10 or more depth-averaged readings taken at two additional cross-sections. Cross-sectional flow measurements were compared to flow calculations derived from conservative tracer data to obtain the best possible estimate. The substrate was quantified with a minimum 300 particle pebble count, composed of 100 measured particles per habitat unit type using a gravelometer and sampling grid (Bunte & Abt 2001a).

3.6.4 Tracer Injection

Injection and measurement of the conservative and non-conservative tracer BTCs required a rigorous injection methodology. Special care was taken to eliminate cross contamination of injectant and sampling equipment in the laboratory and field, as the injection solution is on the order of 10,000 times the concentration of the sampled stream water. Additionally, all sampling equipment was sanitized in a four-step acid wash process. Plateau NO_3^- levels were targeted at four times the ambient concentration and stream Br^- concentration was targeted to be at least 2 mg/L, but not to exceed 5 mg/L. All injection solutions were well mixed and below the solubility limits of the combined solutes.

Tracers were injected at least 20 m upstream of the first transect and were mixed via riffle or flow constriction within that distance. A 60 min pulse of an aqueous potassium nitrate (KNO_3) and sodium bromide (NaBr) solution was injected into the stream using a Watson-Marlow Model 323S/D peristaltic pump (Watson-Marlow Inc., Wilmington, Massachusetts, USA), powered by a deep cycle 12 V battery and inverter. Injectant breakthrough curves were collected by hand at both the upstream and downstream ends of each study reach. These filtered 20 ml breakthrough curve samples were collected at 2 min intervals for 10 min before and 130 min after the injection period, for a total sampling time of 200 min. All samples were then kept on ice in the field and frozen of the day of collection. Streambed disturbance was minimized at both sampling locations.

3.6.5 Benthic Organic Matter Collection

Benthic organic matter (BOM) on and within the uppermost 5 cm of substrate was collected at nine locations per reach, randomly selected both among repeating habitat units and across selected transects. To sample, BOM sections of streambed were isolated by sinking a 0.262 m diameter cylinder into the substrate. All organic matter on the bed surface was removed before the upper 5 cm of the substrate was agitated to release entrained BOM. Finally, all BOM was removed and screened through a 500 μm sieve mounted to a sampling bucket. Material retained on the sieve comprised the coarse benthic organic matter (CBOM) sample and the portion within the bucket formed the fine benthic organic matter (FBOM) sample. The volume of the FBOM sample was recorded and a representative sub-sample of FBOM was removed for laboratory analysis.

3.6.6 Metabolism

Metabolic activity was measured with an In-situ Troll 9000 Multi-parameter Probe (In-Situ Inc. Fort Collins, Colorado, USA) equipped with optical dissolved oxygen

(DO), temperature, and barometric pressure sensors. Probes were deployed immediately following the tracer injection and logged ambient conditions at 10 min intervals for 48 hrs. A vented cable was used to equalize the unit to atmospheric pressure, eliminating post-processing pressure correction.

3.6.7 Data and Sample Analysis

All survey data were post-processed with Microsoft Excel® spreadsheets (Microsoft, Redmond, Washington, USA), converting 3-D survey points into cross-sectional data. These data were then used to estimate at-a-station and downstream channel characteristics. Intermediate survey points were used as the root of additional cross-sections, linearly interpolated from the remaining cross-sectional information. Habitat units were then linked to the various geometric, textural, and hydraulic parameters which described them. Reach-averaged geometric, hydraulic, and textural parameters were proportionally upscaled from the average condition of each habitat unit using the areal distribution of each unit.

Geomorphic complexity metrics selected and developed for this study focused on spatial scales between 1 m^2 and the entire reach. Dimensionless metrics that are largely scale invariant were developed due to their applicability to streams of any size. Finally, metrics developed for equally spaced field surveys were modified to adapt to unequally spaced data points. (Table 3.4 provides a summary of currently published geomorphic complexity metrics, while Table 3.5 provides a listing of metrics developed specifically for this project.)

BOM samples were analyzed for their ash-free dry mass (AFDM) content. FBOM samples were filtered through 1.6 μm pore size 9 cm diameter Whatman GFA glass fiber filters, dried at 100°C for 12 hr and oxidized in a muffle furnace at 500°C for 6 hr. CBOM samples, not requiring a filtering stage, were similarly dried, weighed, and combusted. Both techniques are in accordance with the *Standard Methods for the Examination of Water and Wastewater* (American Public Health Association (APHA) 1998).

BTC samples were stored in a deep freezer from the day of collection until they were thawed immediately before analysis. Analysis for bromide and nitrate concentration was performed on a Metrohm Ion Analysis ion chromatograph (IC) (Metrohm, Herisau, Switzerland) following USEPA Method 300.0 (USEPA 1993). Quality assurance included 10% duplicate samples and recalibration of the instrument every 200 samples and/or when standards fell outside of the 10% accuracy limit.

Table 3.4 – Published metrics of sediment attributes and complexity considered in this study

Group	Metric	Units	Equation	Description
Substrate Variation and Bed Permeability	Fredle number (Lotspeich & Everest 1981)	L	$f_i = \frac{\sqrt{d_{16}d_{84}}}{\sqrt{\frac{d_{75}}{d_{25}}}}$	The Fredle index was developed as a measure of bed porosity with respect to salmonid spawning beds; in gravel-bed streams, as the numerator (the geometric mean) increases and denominator (a sorting coefficient) decreases, the porosity of the bed generally increases.
	Geometric sorting (Bunte & Abt 2001b)	-	$\sigma_g = \sqrt{\frac{d_{84}}{d_{16}}}$	Sorting is an expression of the standard deviation of the distribution used to measure the scatter within a sample; a high degree of sorting results in a low value whereas a sample with a wide range of observed sizes would be poorly sorted and have a high sorting value.
	Gradation coefficient (Bunte & Abt 2001b)		$s_{grad} = \frac{\left(\frac{d_{84}}{d_{50}} + \frac{d_{50}}{d_{16}}\right)}{2}$	The gradation coefficient computes the spread of a sediment distribution, hence, sediment with a large gradation coefficient has a wide range of particles and is said in the engineering world to be well graded.
Longitudinal Variation	Average water-surface concavity (Anderson *et al.* 2005)	-	$AWSC = \left(\frac{1}{n}\right)\left(\sum_{i=1}^{n}\left\|\frac{d^2z_S}{dx_i^2}\right\|\right)$	Concavity at successive points along the water surface profile was calculated as the second derivative of water surface elevation as a function of distance downstream.
	Sinuosity	-	$S = \frac{L}{L_v}$	Ratio of the length of stream (L) to the straight line valley length (L_v).
	Longitudinal roughness (Gooseff *et al.* 2007)	L	$\varepsilon = \frac{\sum_{i=1}^{n}\left[\left(z_{obs,i} - z_{p,i}\right)\right]}{n}$	This function determines the average deviation of the thalweg elevation from a straight line approximation from the upstream to downstream end of the channel, suspect to large values for concave up or down channels.
Patch-scale Variation	Number of geomorphic units	-	(none)	Count of total designated geomorphic units within the reach.
Multi-metric Indices	Metric of complexity (Gooseff *et al.* 2007)	L	$\chi = S_o \cdot S \cdot \varepsilon$	Slope (S_o) provides for the potential for variability in pressure head distributions, sinuosity (S) provides the potential for lateral hydraulic complexity, and the longitudinal roughness (ε) characterizes longitudinal variability.

Table 3.5 – Metrics of complexity developed for this project

Group	Metric	Units	Equation	Description
Substrate Variation and Bed Permeability	Sediment coefficient of variation	L^{-1}	$CV_s = \frac{\sqrt{\frac{d_{84}}{d_{16}}}}{d_{50}}$	The sediment coefficient of variation measures the ratio of the geometric standard deviation to the mean particle size in the distribution.
Longitudinal Variation	Longitudinal roughness[a]	-	$LR = \frac{\sum_{i=1}^{n}\left[\left(z_{obs,i} - z_{p,i}\right)\cdot l_p\right]}{D}$	Instead of equally weighting each point, each deviation value is weighted by the proportion of the reach it best represents, and then the total values are summed and non-dimensionalized by dividing by hydraulic depth.
	Longitudinal variation	-	$TV = \frac{L}{D}\left(\sum_{i=2}^{n}\frac{1}{x_i - x_{i-1}}\cdot\left\lvert\left(z_{p,i-1} - z_{p,i}\right) - \left(z_{i-1} - z_i\right)\right\rvert\right)$	Composites the variation in the thalweg elevation at adjacent cross-sections, scaled by the inverse of the proportional length of channel, and normalized to the average bed slope (so as to not bias toward steeper channels). Non-dimensionalized by dividing by hydraulic depth.
	Average water surface concavity	-	$AWSC = \left(\sum_{i=1}^{n}\left\lvert\frac{d^2 z_{wi}}{dx_i^2}\right\rvert\cdot l_p\right)$	Concavity at successive points along the water surface profile was calculated as the second derivative of water surface elevation as a function of distance downstream, scaled by the influence of each point; as points were unequally spaced.
	Average thalweg concavity	-	$AThC = \left(\sum_{i=1}^{n}\left\lvert\frac{d^2 z_i}{dx_i^2}\right\rvert\cdot l_p\right)$	Concavity at successive points along the thalweg of the channel, calculated as the second derivative of thalweg elevation as a function of distance downstream, scaled by the influence of each point; as points were unequally spaced.
Cross-sectional/ Width Variation	Width residual[a]	-	$WR = \frac{\sum_{i=1}^{n}\left[\left\lvert w_{w,i} - \overline{w}\right\rvert\cdot l_p\right]}{\overline{w}}$	Proportionally weighted difference between the current and average width summed up for the entire channel.
	Width variation	-	$WV = \frac{L}{\overline{w}}\left(\sum_{i=2}^{n}\frac{1}{x_i - x_{i-1}}\cdot\left\lvert w_i - w_{i-1}\right\rvert\right)$	Takes into account the variation in wetted width (w) between adjacent points, weighted by the proportion of the reach represented. Non-dimensionalized by dividing by average width.

[a] Proportion of influence (l_p) is a measure from half way between the point of interest and the next point upstream, to half way between the point of interest and the next point downstream, $l_p = \frac{\left(x_i - x_{i-1}\right)/2 + \left(x_{i+1} - x_i\right)/2}{L}$.

3.6.8 BTC Modeling

The One-dimensional Transport with Inflow and Storage (OTIS) model (Runkel 1998) was used to model BTCs for both the conservative and non-conservative tracers using a transport based approach (Runkel 2007). OTIS employs the advection-dispersion equation, with additional consideration of transient storage and first-order decay, to define a BTC based on a set group of parameters (Equations (3.3a-b)). This suite of equations is solved using a Crank-Nicolson finite difference algorithm:

Transient Storage Model (TSM) employed by OTIS

Equation (3.3a): Spatial and temporal variation in solute concentrations in the main channel

$$\frac{\partial C}{\partial t} = -\frac{Q}{A}\frac{\partial C}{\partial x} + \frac{1}{A}\frac{\partial}{\partial x}\left(AD\frac{\partial C}{\partial x}\right) + \frac{q_L}{A}(C_S - C) + \alpha(C_S - C) - \lambda C$$

Equation (3.3b): Spatial and temporal variation in solute concentrations in the transient storage zone

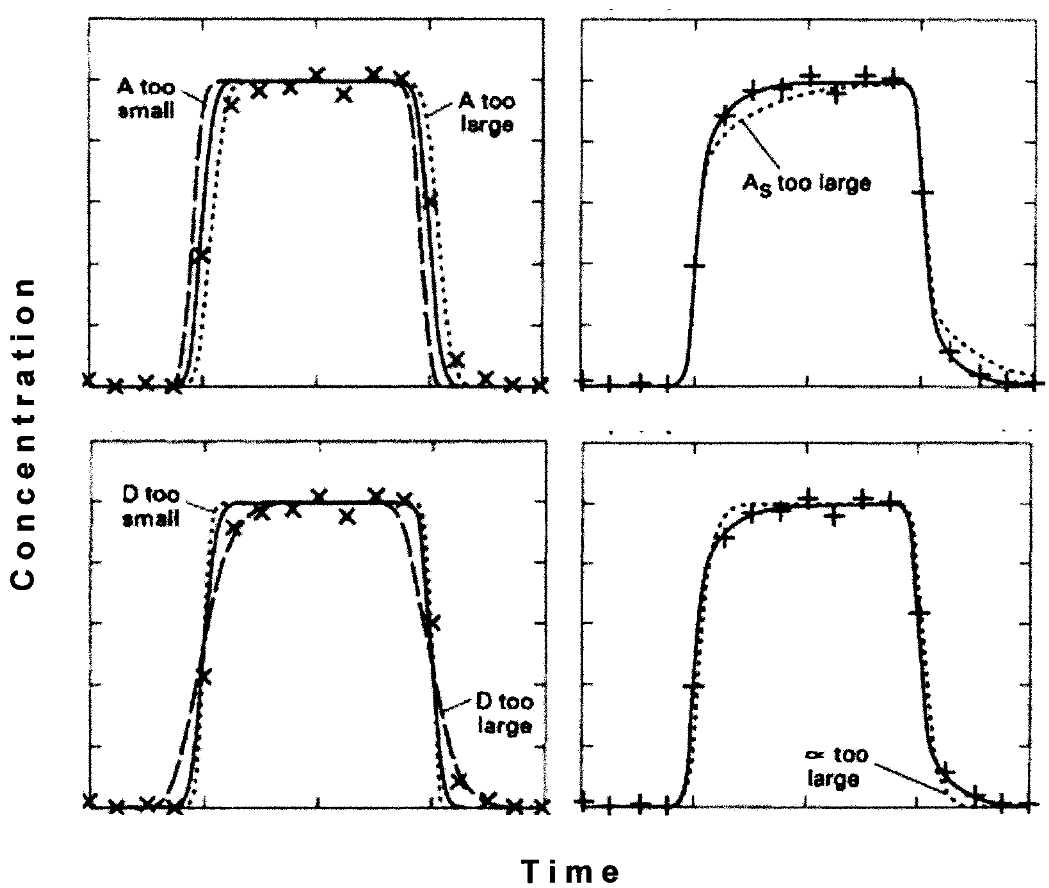
$$\frac{\partial C_S}{\partial t} = \alpha\frac{A}{A_S}(C - C_S) - \lambda_S C_s$$

(Most of the definitions for variables can be found in Table 3.6, the rest can be found in Section 3.11.)

Table 3.6 – Definition of symbols used in text and equations

Parameter	Units	Definition
A	m^2	main channel cross-sectional area
A_s	m^2	storage zone cross-sectional area
C	mg/L	solute concentration
C_S	mg/L	solute concentration in transient storage zones
D	m^2/s	longitudinal dispersion coefficient
Dal	-	index of the influence of transient storage on reach-scale advection transport
d_{16}	mm	16th percentile of sediment size distribution, one standard deviation below the mean
d_{25}	mm	25th percentile of sediment size distribution
d_{50}	mm	50th percentile of sediment size distribution, mean particle size
d_{75}	mm	75th percentile of sediment size distribution
d_{84}	mm	84th percentile of sediment size distribution, one standard deviation above the mean
F_{med}^{200}	-	fraction of median travel time due to transient storage, normalized for a 200 m reach
h	m	water column depth
LR	m	longitudinal roughness
P	m	wetted perimeter
q_L	m^2/s	lateral inflow per unit length
R	m	hydraulic radius
s_grad	-	sediment gradation coefficient
S_w	m	nutrient uptake length: average distance of stream a distance traveled in dissolved form
S_o	-	water surface slope
t	s	time
U	$\mu g/m^2s$	uptake flux: solute uptake rate per unit area of stream bottom
u	m/s	cross-sectional-averaged stream velocity
v_f	mm/s	nutrient uptake velocity: mass transfer coefficient from water to benthic compartment
x	m	downstream distance
x_i	m	downstream distance from previous measurement
z_{obs}	m	observed thalweg elevation
z_p	m	predicted thalweg elevation from straight line approximation
z_{wi}	m	observed water surface elevation
α	1/s	storage zone exchange coefficient
λ	1/s	main channel first-order decay coefficient
λ_{eff}	1/s	effective uptake coefficient: sum total of the corrected observed uptake coefficient λ_s (for hydrologic effects of transient storage) and the main channel uptake coefficient
λ_s	1/s	storage zone first-order decay coefficient

OTIS best fits a parameterized curve to observed conservative tracer values by optimizing the transient storage parameters, channel area (A), area of storage (A_S), dispersion coefficient (D), and exchange coefficient (α)). Of the four transient storage variables, A is the easiest to estimate as it is a function of the magnitude and arrival time of the downstream BTC. The other variables, A_S, D, and α, are based on subtle changes in the curvature of the shoulders of the BTC, and thus they are more difficult to parameterize and contain more inherent uncertainty. The nutrient uptake parameters are similarly fit to the observed BTC of the non-conservative tracer.



Note: Measured concentrations are designated with an x or +, while the solid curve shows an idealized model fit. Dotted and dashed lines show the effect of under or over estimating the designated parameters.

Figure 3.6 – Hypothetical OTIS modeling of a conservative tracer to BTC data (Stream Solute Workshop 1990).

While the OTIS software package does offer parameter optimization in the companion program OTIS-P, early model runs revealed challenges in model convergence and limited parameter uncertainty data; therefore a more robust and descriptive inverse modeling tool was sought for this project. The computer code for universal inverse modeling, UCODE (Poeter & Hill 1999), was selected to parameterize each OTIS model due to its extensive convergence controls, sensitivity analysis, and model fit statistics. Developed for nearly any text input numerical model, UCODE has been extensively used in groundwater modeling with MODFLOW and successfully used in conjunction with OTIS (Scott *et al.* 2003, Briggs *et al.* 2009). (See Figure 3.7 for a flowchart of the interaction between UCODE and OTIS.) UCODE is optimized for non-linear regression by minimizing a weighted least-squares objective function. For this project, the optional double-dogleg trust region approach outperformed the default modified Gauss-Newton method for model convergence and stability (Poeter *et al.* 2005).

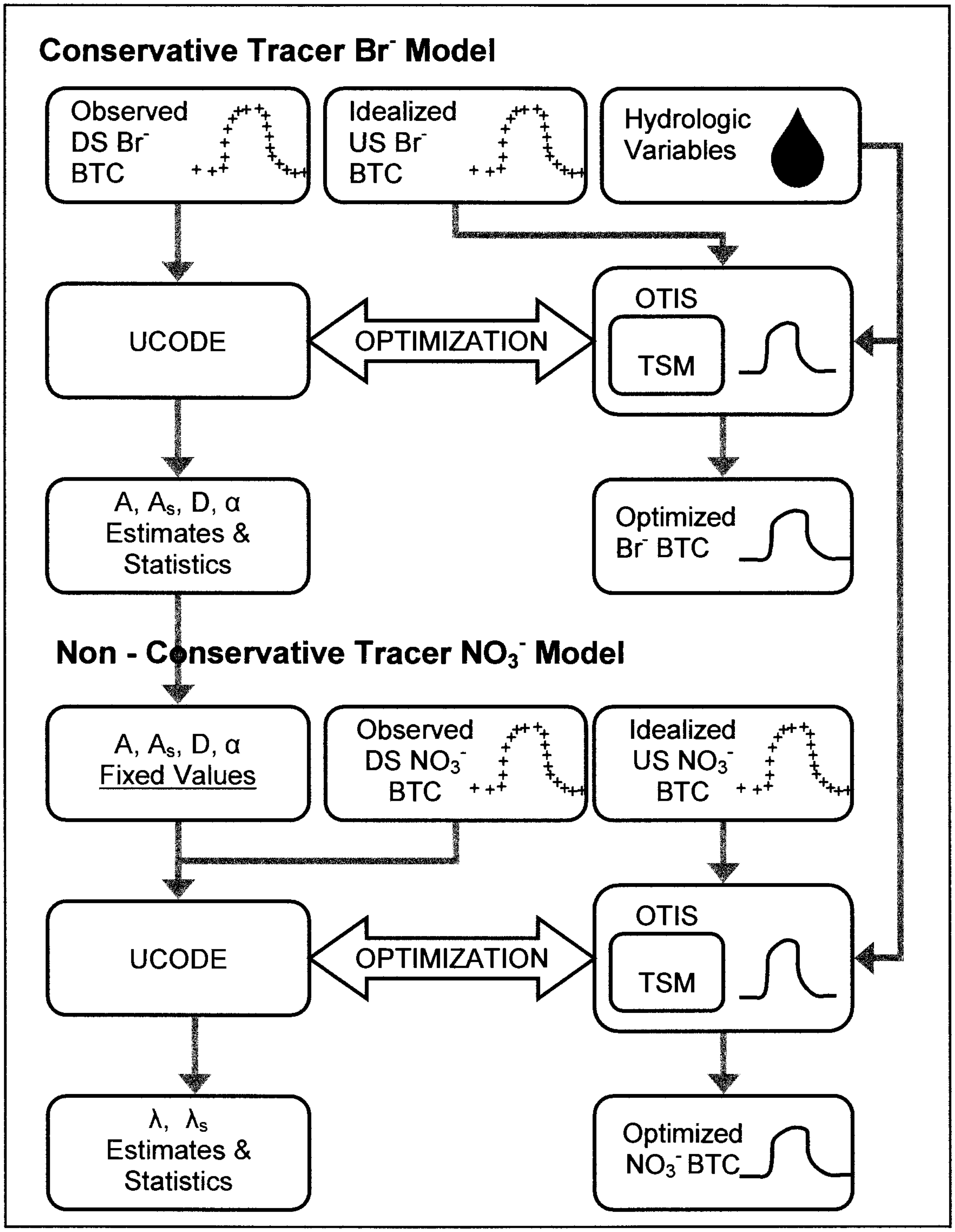


Figure 3.7 – OTIS-UCODE modeling flowchart

UCODE calculates fit independent statistics using parameter sensitivities, i.e. measures of the potential of an observation to make a difference in the model as opposed to the actual influence. Parameter sensitivity is the scale at which model predictions change when the value of a particular input parameter is changed. Thus, an array of sensitivities is calculated for the effect of perturbation of each parameter value for each observation. Incorporating the weighting level for each observation, dimensionless scaled sensitivities (DSS) indicate the amount the simulated value would change given a 1% increase in parameter value. DSS values for each parameter are collected into composite scaled sensitivities (CSS), which are the total amount of information provided by all observations for the estimation of one parameter. CSS values clearly show that a parameter can be estimated only if the information contained in the observations, as expressed via their sensitivities, eclipses the effects of the noise in the observational data (Hill & Tiedeman 2007).

UCODE models converge when fractional changes in parameter values are less than 1% between successive iterations (Poeter *et al.* 2005). Model convergence, while necessary for parameter estimation, does not imply finalized parameter estimates. In this study, further scrutiny of parameter estimates was based on four criteria:

1. coefficients of variation (CV) having a value less than 1.0 (i.e., the standard deviation of each parameter being less than the mean estimated value);
2. parameter CSS values greater than 0.1 and the minimum parameter CSS value no less than 1% of the maximum value (Poeter *et al.* 2005);
3. minimal correlation among parameters in model; Hill & Tiedeman (2007) give a guideline that if absolute values of all correlation coefficients < 0.95, then all parameters should be estimated uniquely; and

4. parameter values falling within logical ranges of values (i.e., dispersion rates outside of published values or storage zone areas more than two orders of magnitude larger than the channel area).

3.6.8.1 Modeling Phases

Nutrient uptake modeling using OTIS has typically been performed in a two-stage process (Scott *et al.* 2003, Wondzell 2006, Gooseff & McGlynn 2005). First the conservative tracer BTC is modeled, resulting in parameter estimates related to transient storage, A, A_S, D, and α. Second, the non-conservative tracer BTC is fit to a model using fixed values for the transient storage parameters found from the conservative tracer. This second model computes the nutrient uptake parameters, in-channel uptake (λ) and storage zone uptake (λ_s).

Due to larger than expected uncertainty in parameter values and the inability to get some models to converge using the two-stage process, I developed an innovative approach to couple the conservative and non-conservative models into a single combination model that used the conservative BTC to estimate parameter values to A, A_s, D, and α; and the non-conservative BTC to help estimate A, A_s, D, and α plus the uptake parameters of λ and λ_s. The two models were loosely coupled, with the conservative and non-conservative models run back-to-back within each model iteration. Multiple input curves have been often used in groundwater modeling with UCODE (E. Poeter, pers. comm. 2008), but to my knowledge, this is the first time the technique has been performed in conjunction with the OTIS model. Statistical comparison of the parameters garnered from the two-stage model and combination model revealed no significant improvement in either parameter estimates or number of successful models, thus I was unable to conclude whether a combination model holds any advantage over the two-stage model.

3.6.8.2 Variance Analysis

To mediate the effects of sample variation on the performance and stability of the OTIS-UCODE models, the following techniques were used:

1. Observations greater than three standard deviations from the modeled curve were trimmed from the data set in an iterative process. Each new trimmed data set was reevaluated until all points lay within the 3 standard deviation envelope. A few models required at most two iterations of this process, while over one-half of the models did not have any points outside of the 3 standard deviation envelope to begin with.
2. Refinement of BTC by a three-point moving median filter, effectively minimizing sampling noise and diminishing the effect of outliers, while maintaining the fundamental shape of the BTC including the preservation of signal edges and stepwise discontinuities (Tukey 1977).
3. Additional techniques of model manipulation included:
    a. iteratively estimating one parameter at a time (to "walk" the model toward acceptable parameters),
    b. using mathematically relative relationships between parameters to keep highly uncertain parameters stable, and

c. if all else failed, fixing the value for flow area equal to that of the reach-averaged field measured quantity.

The OTIS model requires an upstream tracer BTC that is modified in relation to the best-fit transient storage or nutrient uptake parameters to match the downstream BTC. A simplified orthogonal pulse (known from the injection pump operation) can be used, but more representative parameters can be obtained from the model if an actual upstream BTC is measured. Due to delays and difficulties with sample processing, I was only able to obtain upstream BTCs for 5 of the 12 injections. For consistency among models it was determined to use best-fit upstream BTCs, derived from other injections at the same site. If the available data showed little shift in the shape of the BTC between the injection point and the upstream end of the study reach, an orthogonal pulse was applied. To better establish the timing of the upstream BTC, the modeled area was matched to the cross-sectional average surveyed value.

3.6.8.3 Conversion of Time-series Parameters to Steady-state Values

A primary advantage of time-series analysis is the differentiation between in-channel uptake (λ) and storage zone uptake (λ_s). To provide comparability between time-series and steady-state studies, λ and λ_s are combined into an effective uptake rate (λ_{eff}) or the sum of the in-channel uptake rate and a corrected storage zone uptake rate (Equation (3.4)):

Equation (3.4): Effective uptake rate

$$\lambda_{eff} = \lambda + \frac{\alpha \lambda_s A_s}{\alpha A + \lambda_s A_s}$$

The corrected storage zone rate moderates the level of storage zone uptake by taking into account the hydrologic effects of (1) water from the main channel needing to pass into the storage zone before it can be removed and (2) the impact of λ_s on nutrient mass as a function of residence time in the storage zone. Cross comparable metrics can then be calculated (Runkel 2007) as shown in Equations (3.5) through (3.7):

Equation (3.5): Uptake length

$$S_w = \frac{\text{velocity}}{\lambda_{eff}}$$

Equation (3.6): Uptake velocity

$$v_f = \lambda_{eff} \cdot \text{depth}$$

Equation (3.7): Areal uptake rate

$$U = v_f \cdot \text{concentration}$$

3.6.9 Metabolism Modeling

Metabolic activity is represented by the gross primary productivity (GPP) of a reach. GPP is the rate at which the producers within each reach capture and store chemical energy and respiration (R) is rate of cellular consumption by living material. The difference between the two is expressed as net primary productivity (NPP = GPP – R). While denitrification is a process of respiration, the majority of assimilation is a result of primary productivity.

The Stream Metabolism Program (SMP), developed by the USGS National Water Quality Assessment Program and based on the original work of Odum (1956), was used to model whole stream metabolism (Bales & Nardi 2007). Model input data included stream discharge, dissolved oxygen concentration (measured continuously for at least 30 hrs), water temperature, and barometric pressure. The single station dissolved oxygen application of SMP was used to calculate the GPP of each reach, or the amount of oxygen used per square meter of streambed per day, corrected for respiration.

3.6.10 Statistics

Unconstrained ordination of channel complexity variables was performed with principal component analysis (PCA) to assess variable information content and redundancy (Jolliffe 2002). The most informative variables were then compared across study reaches to examine variation in channel complexity. PCA axis scores were not used in subsequent analyses, but the information from the ordination was used to help select variables for inclusion in regression analyses of transient storage and nutrient uptake processes. Comparisons of complexity metrics were performed graphically.

A multi-step statistical process quantified connections among the independent channel measurements and the dependent transient storage and nutrient uptake parameters. First, the data set of complexity, substrate, channel, BOM, metabolism, and hydraulic measures, was reduced to a set of independent variables using PCA and correlation analysis. Next, separate data sets were tiered based on *a priori* knowledge of the influences of transient storage and nutrient uptake processes (Table 3.7 and Table 3.8, respectively). Each reduced independent variable set was then regressed against either the transient storage or nutrient uptake metrics. Both general linear and power models were examined, with power models proving to be more robust. The multiple regression models of two variables or less, sorted by their adjusted R^2 value, and subject to passing parameter and overall model significance at a level of α = 0.10, were examined for consistent patterns of variable inclusion and influence direction. Only then were the most significant and interpretable models selected. All statistical tests where performed using SAS 9.2 (2008, SAS Institute, Inc., Cary, North Carolina, USA).

Table 3.7 – Process-based description of variables included in transient storage best subsets model

Variable Name	Model Name	Comments
Reynolds number	Re	A ratio of the product of depth and velocity to viscous forces, the Reynolds number factors in both head pressure and driving energy that could influence the hyporheic component of transient storage
Specific stream power	ω	Factoring in both slope and unit flow rate, specific stream power relates the flow energy per unit area of a stream channel
Longitudinal roughness	LR	Measurement of the longitudinal deviation from a constant slope channel
Width residual	WR	A new metric derived for this study, width residual is a measurement of the width variation between adjacent transects
Average thalweg concavity	AThC	Measures the reach-averaged change in slope of the deepest portion of the cross-section, a substitute for AWSC, which was not used due to high correlation with other variables
Sediment geometric coefficient of gradation	s_grad	The higher the value of gradation coefficient, the greater the variance of the sediment distribution; the lower, the more uniform the distribution
Relative submergence	R/d_{84}	A ratio of the hydraul^ic radius over the 84th percentile grain size, relative submergence relates the relative grain roughness of the channel and is also a relative (dimensionless) measure of flow depth

Table 3.8 – Process-based description of variables included in nutrient uptake best subsets model

Variable	Model Name	Comments
Gross primary production	GPP	The higher the metabolic rate of the system, the more nutrients that should be taken up.
Fraction of median travel time	F^{200}_{med}	A measure of transient storage which increases the exposure to the biologically-rich hyporheic zone and increasing residence time within in-channel compartments.
Ratio of storage to channel area	A_s/A	Simple transient storage metric that relates the cross-sectional area of the storage zone (including both in-channel and hyporheic zones) to that of the advective portion of channel.
Total benthic organic matter	TBOM	Vitally important as an energy source for denitrification (one of the multiple forms of NO_3^- uptake).
Reynolds number	Re	Based on the product of depth and velocity, the Reynolds number factors in both head pressure and driving energy that could help promote nutrient uptake.
Relative submergence	R/d_{84}	A ratio of the hydraulic radius over the 84th percentile grain size, relative submergence relates the relative grain roughness of the channel and is also a relative (dimensionless) measure of flow depth.
Ambient concentration of NO_3^-	C_NO3	As NO_3^- levels increase, the efficiency of uptake often declines.

Following with the recommendations of Hanafi *et al.* (2007), cross-site parameter uncertainty was quantified with Monte Carlo simulations. Each OTIS parameter, transient storage metric, and nutrient uptake metric was evaluated. These simulations were based upon the output statistics as estimated by the UCODE two-phase model and simulations for each parameter were based on normally distributed random variation over 10,000 iterations.

3.7 Results

3.7.1 Regression Models

Transient storage and relative submergence were consistently significant predictors of nutrient uptake; F^{200}_{med} is significantly related to both nutrient uptake measures, S_w and v_f (Table 3.9). Relative submergence and longitudinal roughness were the strongest predictors of F^{200}_{med} (p = 0.067). Representative regression models were chosen from all significant models as indictors of the general trend of both transient storage and nutrient uptake (Table 3.9).

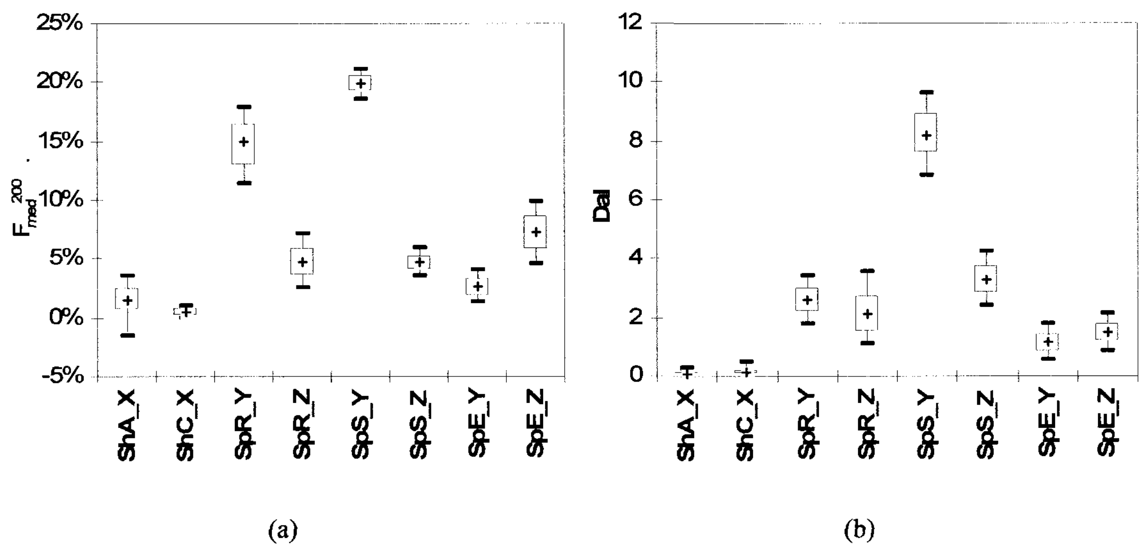
Table 3.9 – Representative regression models

Variable	Sample Model	Model p-value	Adjusted R^2	β_1 P-value	β_s P-value
Fraction of Median Travel Time – F_{med}^{200}	$F_{med}^{200} = 2.78 * (R/d_{84})^{1.15} * LR^{1.96}$	0.067	0.53	0.026	0.086
Uptake Length – S_w	$S_w = 1.06 \times 10^{3} * (F_{med}^{200})^{-0.20} * (R/d_{84})^{-0.23}$	0.007	0.81	0.053	0.050
Uptake Velocity – V_f	$v_f = 6.65 \times 10^{-6} * Re^{0.85} * (F_{med}^{200})^{0.29}$	0.045	0.59	0.023	0.040
	$v_f = 2.26 \times 10^{-7} * Re^{1.067} * (R/d_{84})^{0.38}$	0.057	0.56	0.022	0.051

No significant regression models were found relating GPP, NO_3^- concentration, or any measures of BOM with either nutrient uptake metric (S_w or v_f). Longitudinal roughness was the only geomorphic complexity metric that was a significant predictor of F_{med}^{200} in the regression models.

3.7.2 Transient Storage and Nutrient Uptake Estimates

Monte Carlo simulations indicated significant differences ($p < 0.10$) in the transient storage metric (F_{med}^{200}) between injections Y and Z at each Spring Creek reach. Additionally, F_{med}^{200} was significantly different among Y injections at Spring Creek, but no significant difference was found among the Z injections after the small flood and at a higher flow rate. Additionally, Sheep Creek F_{med}^{200} values were the lowest of all eight injections, though no significant difference was found between Sheep A and Sheep C reaches (Figure 3.8a) Mean values of the Damkohler number ranged from 0.1 to 8.2, all within an order of magnitude of the target value of 1.0 (Wagner & Harvey 1997) (Figure 3.8b).

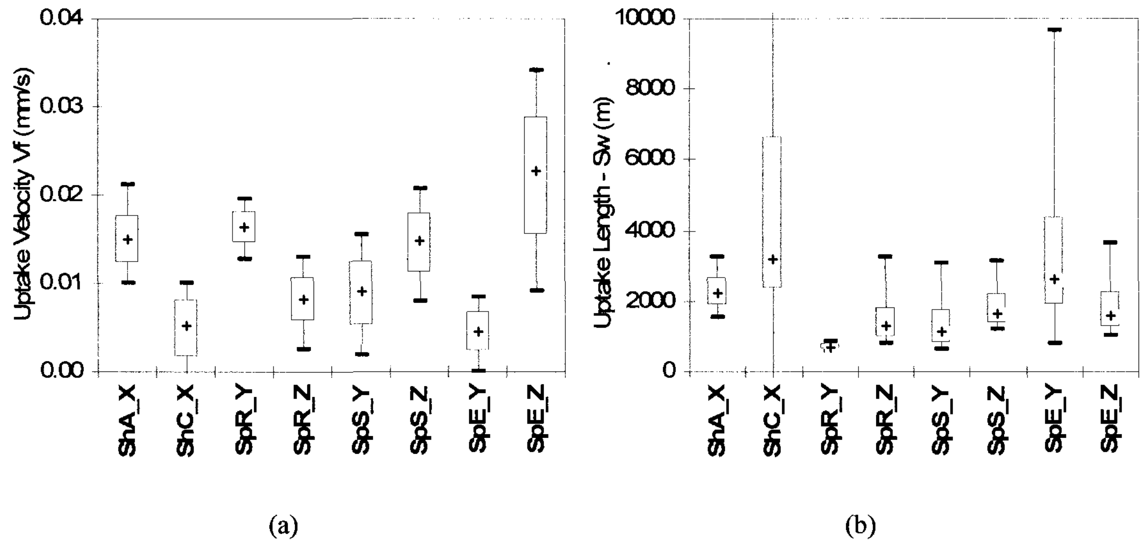


Note: Box plots display Monte Carlo simulation derived 10th, 25th, 50th, 75th, and 90th percentiles

Figure 3.8 – Distributions for (a) the transient storage metric F_{med}^{200} and (b) the index of the influence of transient storage on advection transport DaI across tracer injections

No apparent differences exist between land use types for either reach-scale nutrient uptake metric, though the inter-site differences in v_f at Sheep Creek were similar in magnitude to intra-site differences before and after the high flow event on Spring Creek. Considering individual injections on Spring Creek, v_f was significantly different ($p < 0.10$) between the Y and Z injections at the Edora reach and notably different at the Railroad reach but not at a $p < 0.10$ level. Uptake velocities at Sheep A and Sheep C reaches were significantly different ($p < 0.10$) (Figure 3.9a). With respect to S_w, SpR_Y was significantly different ($p < 0.10$) from ShA, SpE_Z, and SpS_Z. (Figure 3.9b)

(Graphical comparisons of individual OTIS-UCODE parameters are presented in Appendix C.)

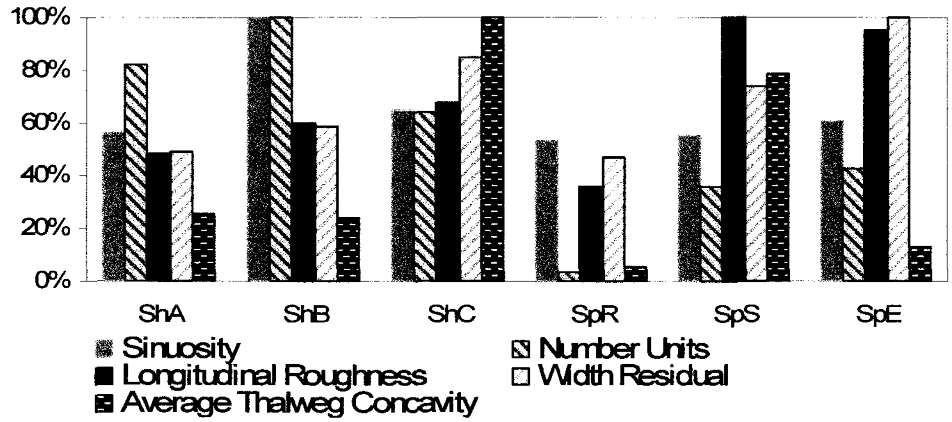


Note: box plots display Monte Carlo simulation derived 10th, 25th, 50th, 75th, and 90th percentiles

Figure 3.9 – Distributions for nutrient uptake metrics (a) v_f and (b) S_w across tracer injections

3.7.3 Geomorphic Complexity

Levels and types of geomorphic complexity vary widely among the six study reaches. The engineered drop structures in the Stuart reach and a complex natural thalweg profile at Edora reach resulted in high measures of longitudinal roughness. With its pool-riffle morphology, Sheep B reach had the greatest width residual and sinuosity. Sheep C reach appeared relatively prismatic, but pools near the upstream and downstream ends and continual small-scale fluctuations in the thalweg profile give rise to the highest average thalweg concavity (Figure 3.10).



Note: All complexity metrics expressed as a percentage of the maximum reach value for the six study sites

Figure 3.10 – Relative geomorphic complexity across study reaches

Graphical comparison indicated substantial differences in levels and types of channel complexity among study reaches. Measures of sinuosity, number of habitat units, longitudinal roughness, width residual, and average thalweg concavity were selected to represent the range of cross-sectional, longitudinal, and planform variations. (A complete table of channel dimensions and geomorphic complexity metrics is provided in Appendix D.)

Correlation among metrics was quite high among the collective metrics of thalweg and width variation and water surface and thalweg concavity (Appendix E). In addition, sediment metrics were highly correlated among themselves, although not with most other measures of physical complexity. PCA analysis of the geomorphic complexity variables indicate that the first eigenvector contained over 50% of the variance and was a composite of nearly half of the variables, suggesting that no one variable or small group of variables contain significantly more unique information than any others. The second eigenvector, containing 21% of the variance, was dominated by measurements of longitudinal and width residuals, which measure the reach-averaged differences from the mean condition (Table 3.10).

Table 3.10 – PCA results of geomorphic complexity metrics

Eigenvector	1	2	3	4
Eigenvalue	11.76	4.91	3.22	1.93
Proportion of Total Variance	0.51	0.21	0.14	0.08
Cumulative Variance	0.51	0.72	0.86	0.95
Water-surface Slope	0.24	-0.13	0.23	-0.09
Darcy's Friction Factor	0.14	-0.08	-0.42	0.27
Sinuosity	0.14	0.26	0.31	0.16
Number of Habitat Units	0.25	0.04	0.22	-0.09
Longitudinal Roughness	0.14	0.30	-0.28	0.14
Thalweg Variation	0.24	-0.11	-0.25	-0.02
Width Variation	0.27	-0.07	-0.13	0.14
Width Residual	0.09	0.41	0.00	0.18
Gooseff's Metric of Complexity	0.28	0.03	0.07	0.10
Average Water Surface Concavity	0.24	-0.20	-0.14	0.10
Average Thalweg Concavity	0.22	-0.06	-0.11	0.29
Dimensionless Longitudinal Roughness	0.04	0.32	-0.02	0.47
Dimensionless Thalweg Variation	0.25	-0.15	-0.11	0.10
Dimensionless Width Variation	0.15	-0.37	-0.09	-0.06
Dimensionless Width Residual	-0.20	0.31	0.08	0.04
Percent Fines	-0.24	-0.17	-0.05	0.16
d_{16}	0.13	0.26	-0.22	-0.38
d_{50}	0.24	0.16	0.05	-0.30
d_{84}	0.27	0.06	0.06	-0.23
Gradation Coefficient	0.12	-0.16	0.43	0.21
Sediment Geometric Standard Deviation	0.16	-0.12	0.39	0.21
Fredle Number	0.26	0.10	0.01	-0.28
Sediment Coefficient of Variation	-0.23	-0.19	-0.02	-0.02

Visual Key

greater than | 0.35 |
greater than | 0.30 |
greater than | 0.25 |
less than |0.20|

3.7.4 Benthic Organic Matter

Benthic organic matter was found in greater quantity in the lower gradient and lower elevation, urban Spring Creek reaches than in the steeper, agricultural Sheep Creek reaches (Figure 3.11). With respect to grazing impacts at Sheep Creek, the reach open to grazing (Sh_C) contained more reach-averaged FBOM and total BOM than the reach within a grazing exclosure (Sh_A), but it contained less CBOM The Stuart reach, characterized by large backwater areas, had the highest level of BOM prior to the small flood. The flood reduced the amount of both fine and coarse benthic organic matter in all three Spring Creek reaches.

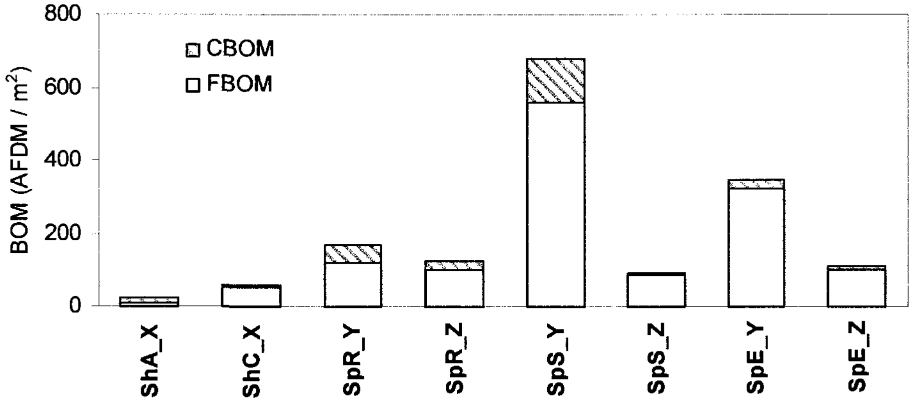


Figure 3.11 – Total benthic organic matter, expressed as the sum total of fine and coarse BOM

3.7.5 Metabolism

GPP was on average higher at the agricultural Sheep Creek reach than it was across the urban reaches of Spring Creek. Additionally, the GPP of the non-grazed Sheep A reach, with dense, shrub-lined banks, was greater than the grazed and open canopy, Sheep C reach (Figure 3.12). Results from the SMP modeling are reported as daily values of gross primary production due to data limitations of a single oxygen profile recorded at the time of each injection.

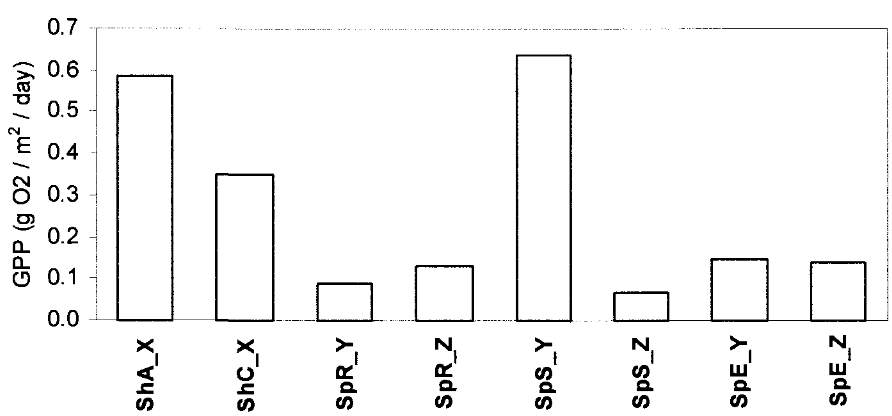


Figure 3.12 – GPP across study injections

3.7.6 OTIS-UCODE Modeling Results

Final model convergence and parameter estimates were obtained for 8 of 12 injections. Graphical OTIS-UCODE model results, including observed data and best-fit model curves for all 12 attempted models, demonstrate the scatter of the observed downstream BTC, the effect of the three-point median, and the fit of the OTIS-UCODE model to the conservative BTC (Figure 3.13) and non-conservative BTC (Figure 3.14). (A table of OTIS parameter estimates, variance, and sensitivity is found in Appendix F.)

Correlation values among transient storage and nutrient uptake model parameters were all less than the acceptable limit of 0.95 (Hill & Tiedeman 2007) (Appendix F). Transient storage models for injections SpR_Z, SpS_Y, and SpS_Z would not converge with all four variables, thus the only term which is also a field measured value, A, was fixed to the reach-averaged physical measurement.

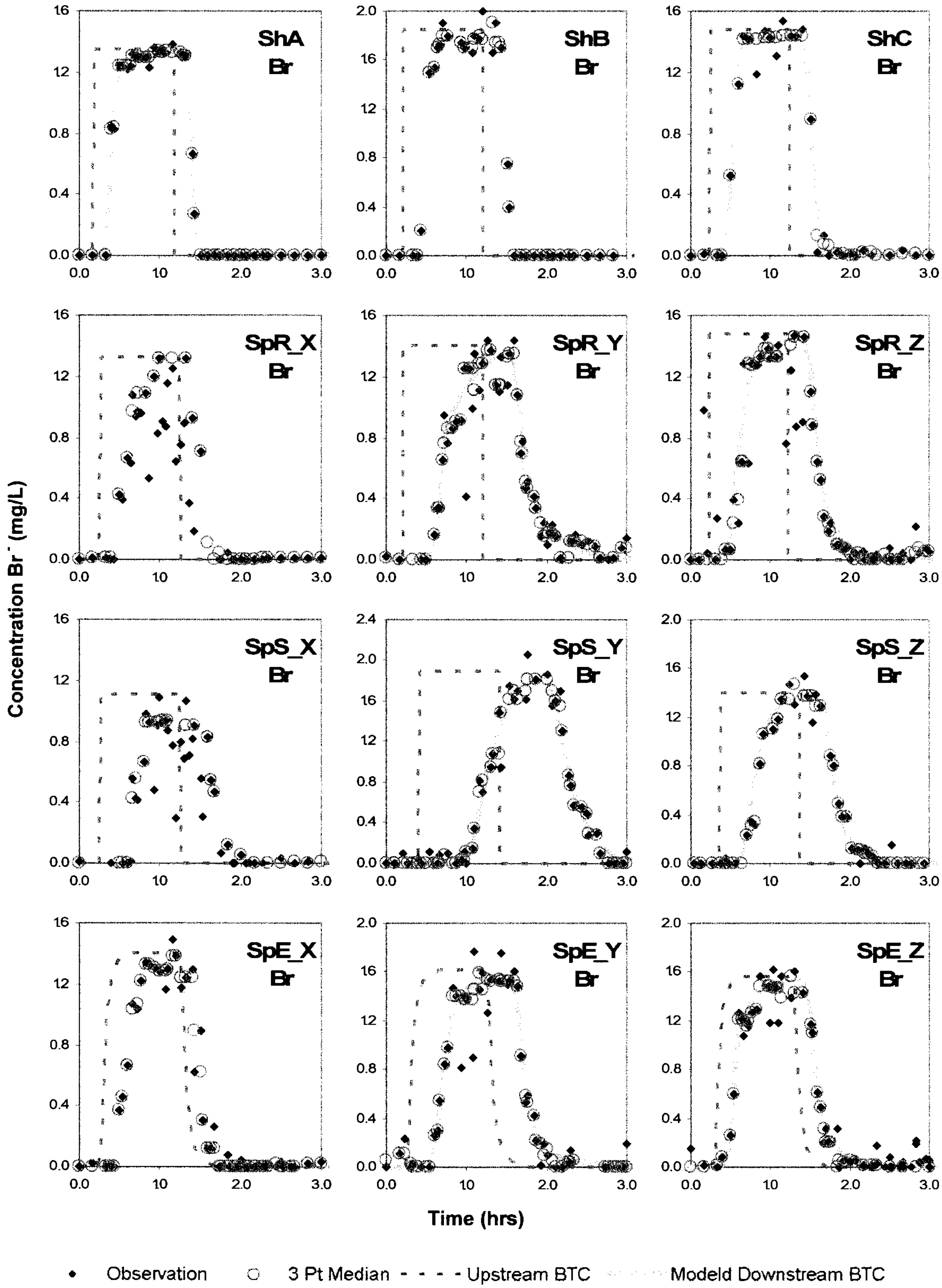


Figure 3.13 – Conservative tracer (Br⁻) OTIS-UCODE modeling BTCs

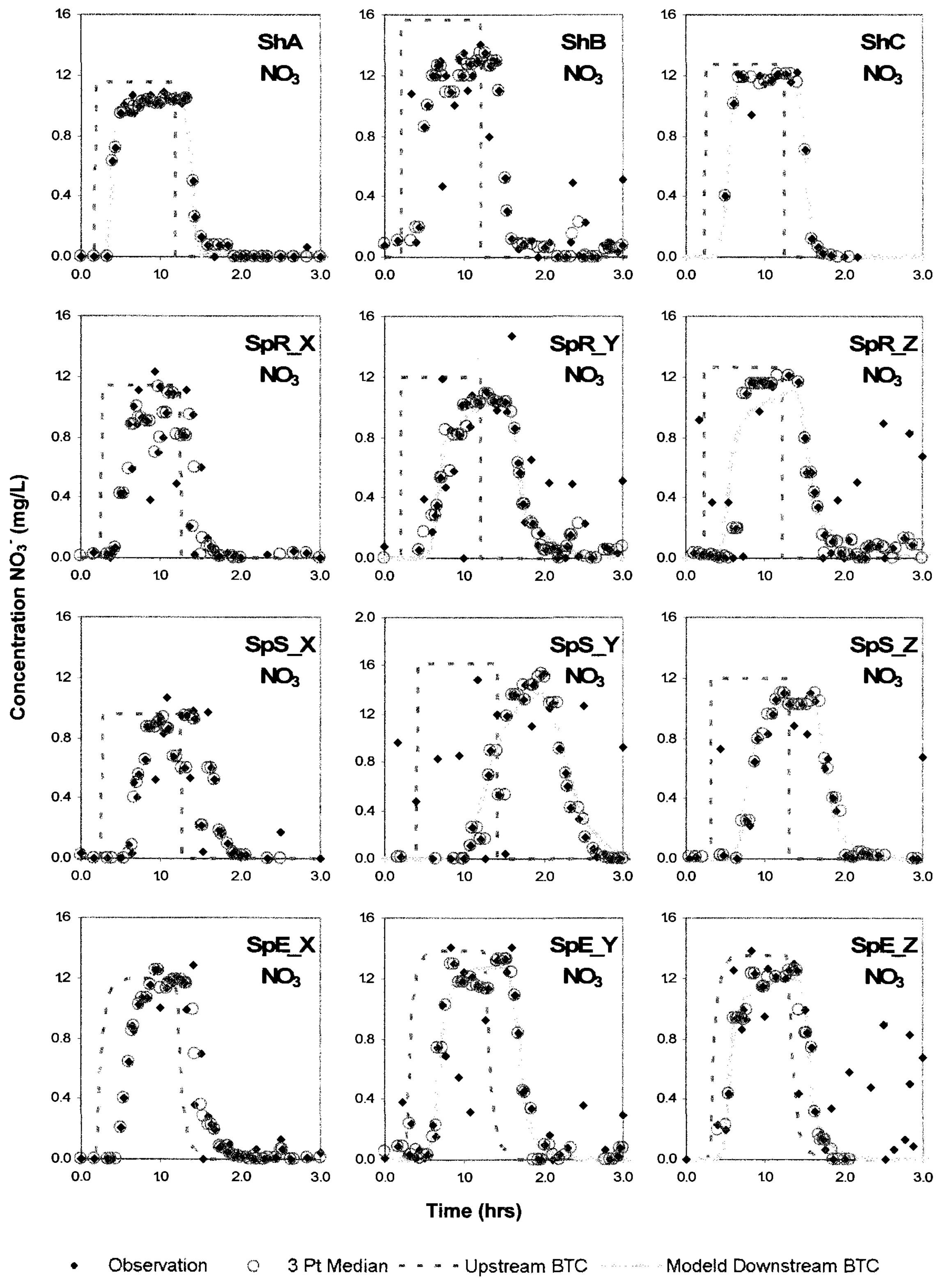


Figure 3.14 – Non-conservative tracer (NO_3^-) OTIS-UCODE modeling BTCs

3.8 Discussion

3.8.1 Influences of Transient Storage and Nutrient Uptake

Significant relationships between both nutrient uptake variables (S_w and v_f) and transient storage support previous findings (Haggard *et al.* 2001, Hall *et al.* In-prep, Valett *et al.* 1996), and a significant dependence on hydraulic factors is consistent with the findings of the first LINX study (Peterson *et al.* 2001). Reynolds number, proportional to the stream unit discharge, was the strongest predictor ($p < 0.025$ in both models) and, with the additional positive relationship between relative submergence and F_{med}^{200}, appears to be driven by stream depth. Multiple studies have found significant positive relationships between v_f and GPP (Mulholland *et al.* 2008b, Mulholland *et al.* 2006, Hall & Tank 2003) and an inverse relationship between v_f and ambient NO_3^- concentration (Mulholland *et al.* 2008b, Payn *et al.* 2005, O'Brien *et al.* 2007), yet neither GPP nor NO_3^- concentration were not significant predictors of uptake in this study. This could be due to the small sample size (n = 8), physical and hydraulic aspects overshadowing biochemical factors, limitations in the one-station modeling of GPP, or a narrow range of ambient NO_3^- concentrations (0.1 to 1.6 mg/L).

The statistical modeling indicates a significant relationship between F_{med}^{200}, longitudinal roughness, and relative submergence. Previous work has suggested linkages between longitudinal roughness and transient storage processes, including significant relationships between mean storage time and a multi-metric variable including longitudinal roughness (Gooseff *et al.* 2007), as well as vertical hydraulic gradients and AWSC (Anderson *et al.* 2005). The inclusion of relative submergence in this relationship further supports the notion that backwater effects are a significant driver of transient storage (Hester & Doyle 2008). Given this connection between backwater and transient storage, and the characteristics of the reaches within this study, it appears that in-stream storage may dominate hyporheic exchange. The three reaches on Spring Creek (comprising six of the eight injections included in the regression model) have limited potential for hyporheic storage as they are underlain by clay-loam soils (Natural Resources Conservation Service (NRCS) *Web Soil Survey* 2009) and contain higher levels of FBOM than any Sheep Creek reach. Furthermore, the heavy vegetation along the channel margin at Railroad reach, backwater from drop structures at Stuart reach, and the natural deep pools and eddies of Edora reach all suggest the existence of in-channel storage. Sheep A and Sheep C reaches had the lowest modeled transient storage values; however, these reaches had the highest slopes and two of the three highest flow rates. Coupling the advective nature of Sheep A and Sheep C reaches (plane bed channels with the highest slopes among study sites) with their minimal values of transient storage provides additional evidence of the predominance of in-channel over hyporheic storage for these sites. Short-term tracer injection experiments are criticized for their inability to track longer spatial and temporal scales of transient storage (Harvey *et al.* 1996), but the clustering of Damkohler values around 1.0 (Wagner & Harvey 1997) and the apparently minimal hyporheic exchange within our study sites indicates the methodology was appropriate to capture the influence of transient storage on reach-scale advective transport.

In the context of results from previous nutrient uptake studies, both v_f and NO_3^- values were found to be within the range of variability of both the LINX II study (Mulholland *et al.* 2008b) and a comprehensive dataset of previous nutrient uptake studies compiled by Tank *et al.* (2008) (Figure 3.15). As geomorphic setting is an important factor in accounting for the variability of uptake rates within the study sites of this project, it is likely that geomorphic context is an important influence on the range of variability in the other datasets as well.

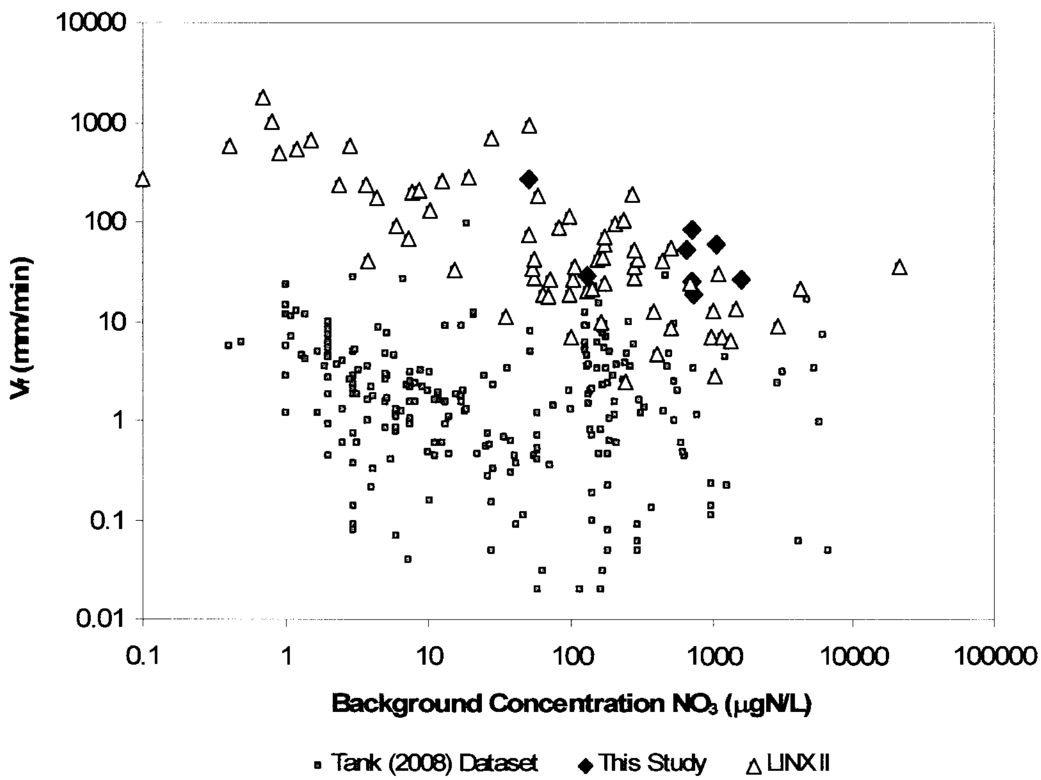


Figure 3.15 – Comparison of v_f and NO_3^- concentration among this study, LINX II (Mulholland *et al.* 2008b), and a dataset of previous uptake studies compiled by Tank *et al.* (2008)

Despite *a priori* notions to the contrary, nutrient uptake does not appear to vary significantly between agricultural and urban sites in either this study or previous studies (Figure 3.15) (Mulholland *et al.* 2008b, Haggard *et al.* 2001). The highly altered urban Spring Creek reaches appear to support nutrient uptake equally as well as the Sheep Creek reaches that are arguably, at least in the case of Sheep A reach, more geomorphically "naturalized" after passive rehabilitation through removal of grazing stressors. The natural recovery of Sheep A reach may be a factor in the greater value for v_f than Sheep C reach, but when contrasted to both Sheep C reach and the urban Spring Creek reaches, Sheep A reach uptake is not substantially different.

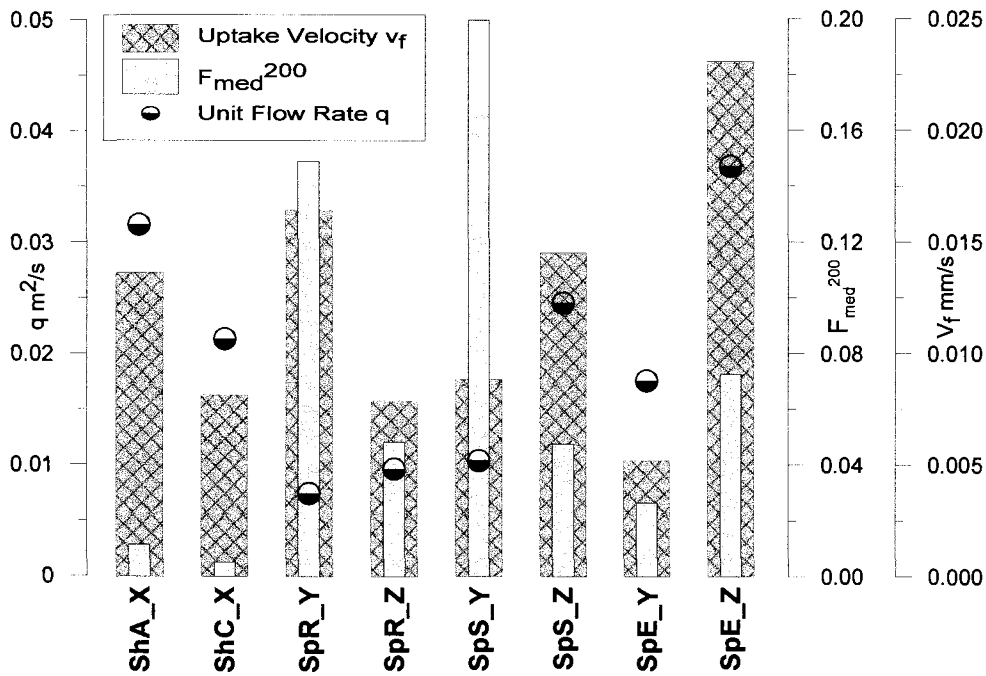


Figure 3.16 – Comparison of F_{med}^{200}, v_f, and unit discharge (q) across study sites

A complex interaction exists between unit discharge and F_{med}^{200} across the eight injections, depending on the geomorphic context, hydrologic history, and level of channel stabilization of each reach (Figure 3.16). The agricultural reaches of Sheep Creek, which were initially hypothesized to be more geomorphically complex and plausibly have more transient storage than the urban reaches of Spring Creek, were found to have much lower levels of overall transient storage, possibly due to the armoring of the streambed resulting from long-duration high flows induced by the upstream reservoir (discussed in more detail later in this section). However, nutrient uptake on Sheep A and Sheep C reaches is not markedly different than the Spring Creek reaches, maybe indicating a differing mechanism of uptake than Spring Creek. The uptake rates in Sheep Creek are plausibly related to the moderate to high levels of GPP which may have been spurred by a disproportionate increase NO_3^- relative to background concentration as compared to the Spring Creek sites. Across all injections, uptake velocity (v_f) scales closely with unit flow rate, as reflected in the regression models.

Few studies have performed repeated tracer injections at the same site to examine the influence of flow level on transient storage (Wondzell 2006, D`Angelo *et al.* 1993) and nutrient uptake (Valett *et al.* 1997). This study explored these relationships with three injections at each Spring Creek reach. Transient storage behaved distinctively at each reach depending on the apparent mode of storage prior to the flood and the effects of the flood on the physical template of the channel. At the Railroad reach, transient storage dropped after the flood. This is possibly due to the flattening of the extensive bank and channel margin grasses by high flows, with bed scour only minimally altering the clay-dominated streambed. Bed and bank stabilization at the Stuart reach limited the effects of the flood on the channel, and the post-flood transient storage decreased as higher flow rates seem to have turned in-channel storage zones into active advective zones. The post-flood Edora reach appeared to retain much of its in-channel storage, but with fines and organic matter swept from the bed, the greater flow may have driven more flow into the hyporheic zone, thus increasing overall transient storage.

The balance of in-channel to hyporheic storage on the Sheep Creek reaches may reflect geomorphic interactions with an altered flow regime. Previous research at Sheep

Creek (Stednick & Fernald 1999, Flenniken *et al.* 2001) and my field observations indicate that operation of Eaton Reservoir, within 2 km immediately upstream of the study reaches, dictates the flow levels year round (Figure 3.17).

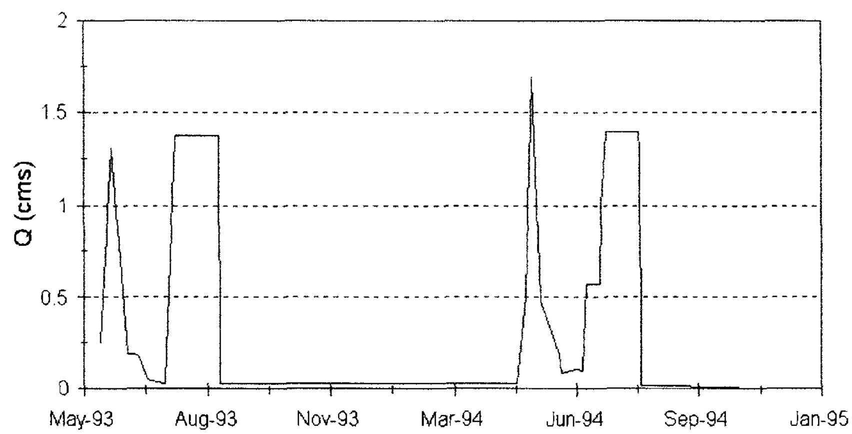


Figure 3.17 – Annual hydrograph for Sheep Creek from Stednick (1999)

In the spring, streamflow releases are minimal until the reservoir is completely filled; only then is water released resulting in the sharp peak in flow around May. During mid-summer, a minimal flow is released, until downstream water rights are invoked and the reservoir is drained with a near-bankfull flow through Sheep Creek. This water release schedule creates an unnatural dual-peaked hydrograph, with the second peak persisting at bankfull flow level for over a month each summer. The apparent effect of this long duration, high flow is to scour Sheep Creek of fines, flush away organic matter, and armor the remaining sediment into a highly interlocked pavement. Flume experiments have demonstrated the same geomorphic response to flow modification as extended duration releases of clear water and limited sediment supply both lead to an armored bed surface (Hassan *et al.* 2006). Combined with the plane bed, steep nature of Sheep A and Sheep C reaches, historical (and current) cattle grazing may have further compacted the streambed, similarly to construction equipment compacting substrate and rendering a "naturalized" looking stream with limited capacity for hyporheic storage (Goetz *et al.* In review). The injection on Sheep B reach, the lowest slope and most sinuous reach on Sheep Creek, could have provided a contrast to the other reaches but unfortunately could not be parameterized in UCODE. In-channel storage has generally shown more influence on nutrient uptake than hyporheic storage in previous studies (Ensign & Doyle 2005, Gucker & Boechat 2004), yet the ability to partition the two transient storage zones (Briggs *et al.* 2009, Salehin *et al.* 2003) would be necessary to examine the distribution and influence of hyporheic and in-channel storage.

Ecological processes are often driven by hydrogeomorphic interactions (Doyle *et al.* 2005), as shown in the complex interplay between transient storage, NO_3^- uptake, and unit flow rate described above. For basic understanding of nutrient uptake processes to evolve, techniques must be developed to better account for complex physical-biologic interactions. Not only is the amount of flow influential, but so is the temporal sequencing of flow events and the geomorphic responses of the channel to those events.

Modeling of nutrient processes has evolved from Eulerian (measures of flux through space) based nutrient budgets to the technique used in this study, applying computer models to translate Lagrangian (particle through time) metrics from Eulerian field measurements (Doyle & Ensign 2009). In an additional step towards the Lagrangian reference system, the use of labeled tracers has provided a technique, though expensive and complex, to compartmentalize the fate of stream nutrients (Peterson *et al.* 2001, Mulholland *et al.* 2008a). This development of a fine scale understanding of nutrient processes is beneficial, but, as demonstrated in this study, future nutrient uptake research must expand its scope and account for geomorphic influences and temporal flow sequences.

Contemporary nitrate uptake research has primarily focused on denitrification, but the LINX II study showed that only a small fraction of NO_3^- removal is due to that process (Mulholland *et al.* 2008b). Thus, hyporheic zone nutrient processing may not be as dominant as previously believed and nutrient uptake within channel storage zones may be of considerable influence in sites similar to those in this study. Accordingly, it may ultimately prove necessary to represent the influence of hydrogeomorphic interactions and flow sequences as documented in this study for upscaling these processes to the drainage network scale.

3.8.2 Geomorphic Complexity

The level of longitudinal, width, substrate, and planform complexity were measured in great detail. Only one metric, longitudinal roughness, was selected by regression analysis as a significant predictor of transient storage, yet results from PCA analysis would indicate that there is a collective influence of many attributes. If stream restoration is to be used as a nutrient management tool (Craig *et al.* 2008), the focus must turn toward restoration of the processes that create and maintain longitudinal roughness via bedforms and other sources of divergence in streambed pressure (Tonina & Buffington 2009). Clearly, multiple forms of complexity are important influences on stream ecosystem functions (Sheldon & Thoms 2006, Brooks *et al.* 2005). As different forms of complexity will influence diverse ecological functions, selection of geomorphic complexity metrics for stream surveys should focus on the scale and processes of interest.

The detailed protocol developed in this study provides a framework of the multiple reach-scale geomorphic complexity metrics available and methods to collect them. Alternatively, experiments controlling flow and geomorphic features could help discern the interactions between geomorphic complexity and transient storage (*e.g.,* Ensign & Doyle 2005), whereas previous flume experiments have focused on applicability of numerical models and subsurface heterogeneity (Zaramella & Packman 2003, Salehin *et al.* 2004).

3.8.3 Sources of BTC Variance

As outlined in Hanafi *et al.* (2007), one of the primary challenges in the OTIS-UCODE modeling was variation in the observed tracer BTCs. This variance could be due to several sources including: (1) inherent variation in hydrologic and biochemical processes, (2) inadequate mixing of tracer solution with stream water, and/or (3) errors associated with sampling and handling of grab samples. The first source was accepted as a consequence of study design; early communication with experts in the field warned of difficulties in measuring uptake in sinuous reaches (B. Peterson, pers. comm. 2006). To examine the second potential source, a comparison of NaCl and Br^- BTCs on the same reaches at similar flows shows much less variability in BTCs collected with *in-situ* specific conductivity sondes than with the Br^- grab samples. This comparison also provides evidence of the third source; grab samples inherently result in higher variability by allowing for small spatial shifts in sampling location and increasing the amount of handling over *in-situ* sampling. Lessons learned from the tracer-injection protocol include: (1) use *in-situ* measurement devices instead of grab samples to increase temporal sampling density, limit handling of samples, and provide immediate results, (2) ensure collection of upstream breakthrough curves, as uncertainty in timing or shape of the upstream BTC leads to modeling difficulty, and (3) if BTCs contain excessive variation, use a systematic signal processing scheme such as a moving median to trim outliers to define the fundamental shape of the BTC.

3.9 Conclusions

Geomorphic setting, including longitudinal roughness and deep tranquil flow, show strong associations with transient storage and multiple measures of nutrient uptake in urban and agricultural streams spanning a gradient of physical modification. The mode and extent of transient storage changes dramatically, likely dependant on the combination of geomorphic characteristics and flow events/magnitudes at each study site. Contributions of this study design include development of a detailed geomorphic reach-scale characterization, a new systematic approach for processing variability in BTCs, exploration of uncertainty in transient storage and nutrient uptake metrics, and investigation of the influence of flow variation through multiple injections at urban stream sites.

Models developed in this study suggest that deeper flow potentially resulting from channel thalweg variability is the primary influence on transient storage and consequently nutrient uptake. Accordingly, channel modification projects should consider the addition of habitat features that will increase in-channel storage for the purpose of additional nutrient removal and benthic organic matter retention, as well as floodplain connectivity and fish habitat diversification.

Future research in this area could be aimed at: (1) further developing techniques to measure transient storage and nutrient uptake at a smaller scale (*e.g.,* Böhlke et al. 2009), (2) supplementary investigation into the apparently strong influence of flow variability and temporal sequences of hydrologic and geomorphic interactions on transient storage and nutrient uptake, (3) investigating the utility, benefits, and limitations of a combination model for estimating nutrient uptake as a possible improvement over the standard two-step model, and (4) expanding techniques to partition transient storage into the components of in-channel storage and hyporheic storage.

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3.11 Symbols, Units, and Abbreviations

Symbols

A	=	main channel cross-sectional area
A_s	=	storage zone cross-sectional area
AThC	=	average thalweg concavity
C	=	solute concentration
C_NO3	=	ambient concentration of NO_3^-
C_S	=	solute concentration in transient storage zones
CV	=	coefficients of variation
concentration	=	ambient NO_3^- concentration
D	=	longitudinal dispersion coefficient
DaI	=	index of the influence of transient storage on reach-scale advection transport
d_{16}	=	16th percentile of sediment size distribution, one standard deviation below the mean
d_{25}	=	25th percentile of sediment size distribution
d_{50}	=	50th percentile of sediment size distribution, mean particle size
d_{75}	=	75th percentile of sediment size distribution
d_{84}	=	84th percentile of sediment size distribution, one standard deviation above the mean
depth	=	hydraulic depth
F_{med}^{200}	=	fraction of median travel time due to transient storage, normalized for a 200 m reach
h	=	water column depth
I_p	=	proportion of influence
L	=	length of stream
L_v	=	valley length
LR	=	longitudinal roughness
p	=	p-value
P	=	wetted perimeter
q	=	unit discharge
Q	=	flow rate
q_L	=	lateral inflow per unit length
R	=	hydraulic radius
R/d_{84}	=	relative submergence

R^2	=	coefficient of determination
Re	=	Reynolds number
s_grad	=	sediment gradation coefficient
S	=	sinuosity
S_o	=	water surface slope
S_w	=	nutrient uptake length: average distance of stream a distance traveled in dissolved form
t	=	time
U	=	uptake flux: solute uptake rate per unit area of stream bottom
u	=	cross-sectional-averaged stream velocity
v_f	=	nutrient uptake velocity: mass-transfer coefficient from water to benthic compartment
velocity	=	cross-sectional average velocity
w	=	width
WR	=	width residual
x	=	downstream distance
x_i	=	downstream distance from previous measurement
X, Y, Z	=	initial injection performed in mid June, a second in late July, and the third in mid-August, respectively
z_{obs}	=	observed thalweg elevation
z_p	=	predicted thalweg elevation from straight line approximation
z_{wi}	=	observed water surface elevation
α	=	storage zone exchange coefficient
ε	=	longitudinal roughness
λ	=	main channel first-order decay coefficient
λ_{eff}	=	effective uptake coefficient
λ_s	=	storage zone first-order decay coefficient
μ	=	optimal value
σ	=	standard deviation
ω	=	specific stream power

Units of Measure

cm	centimeter(s)
°C	degree(s) Celsius
hr(s)	hour(s)
km	kilometer(s)
km^2	kilometer(s)
L	length
L/s	liter(s) per second
m	meter(s)
m/m	meter per meter
m/s	meter(s) per second
m^2	square meter(s)
m^2/s	square meter(s) per second

m^3/s	cubic meter(s) per second
mg/L	milligram(s) per liter
mg/m^2	milligram(s) per square meter
min	minute(s)
ml	milliliter(s)
mm	millimeter(s)
mm/s	millimeter(s) per second
µm	micrometer(s)
$\mu g/m^2s$	microgram(s) per square meter per second
%	percent
s	second(s)
V	Volt(s)

Abbreviations

1-D	one-dimensional
3-D	three-dimensional
APHA	American Public Health Association
AWSC	average water surface concavity
BOM	benthic organic matter
Br^-	bromide
BTC	breakthrough curve
CAREER	Faculty Early Career Development Program
CBOM	coarse benthic organic matter
CSS	composite scaled sensitivities
D	downstream end of reach
DSS	dimensionless scaled sensitivities
DO	dissolved oxygen
FBOM	fine benthic organic matter
GPP	gross primary productivity
IC	ion chromatograph
KNO_3	potassium nitrate
LINX	Lotic Intersite Nitrogen Experiment
MODFLOW	MODular three-dimensional finite-difference groundwater FLOW
N	nitrogen
N_2	nitrogen gas
^{15}N-ammonium	isotopically-labeled ammonium
^{15}N-nitrate, $^{15}NO_3^-$	isotopically-labeled nitrate
NaBr	sodium bromide
NaCl	sodium chloride
NH_4^+	ammonium
NO_3^-	nitrate
NPP	net primary productivity
NRCS	Natural Resources Conservation Service
OTIS	One-dimensional Transport with Inflow and Storage

OTIS-P	predictive component of the OTIS package
PCA	principal components analysis
®	registered
R	respiration
SMP	Stream Metabolism Program
™	Trademark
TSM	transient storage model
U	upstream end of reach
UCODE	universal inverse modeling program
USA	United States of America
USGS	U. S. Geological Survey
USEPA	U. S. Environmental Protection Agency

CHAPTER 4

CONCLUSIONS

This project provides a foundation for future work related to the influence of geomorphic complexity and channel disturbance on transient storage and nutrient uptake, and a better understanding of the influence of diversion dams on downstream hydraulic and sediment conditions. In the first study, flow diversion has been shown to decrease flow velocity, increase fine sediment deposition, and thereby significantly alter benthic habitat. The second study relates geomorphic complexity with transient storage and biogeochemical processing of nitrate in agricultural and urban streams. The results underscore the influence of in-channel storage and flow depth, likely a function of thalweg variability, on transient storage and nutrient uptake. While few of the geomorphic complexity metrics proposed in this project proved significant predictors of either transient storage or nutrient uptake, they could support future studies requiring detailed physical surveys and be adapted to different questions and scales of interest. Collectively, the two studies underscore the primary influence of flow regime on habitat response and nutrient spiraling functions in the context of human influences. In particular, future research in both arenas would benefit from improved descriptions of temporal sequences of flow events interacting with geomorphic context.

APPENDIX A

GENERAL STUDY REACH CONDITIONS

Table A.1 – General study reach conditions

Visit Code	Above/ Below Diversion	Flow Rate, Q (L/s)	Bed Slope, S_o (m/m)	Bed-surface Area (m^2)	Wetted Width (m)	Average Sediment Size, d_{50} (mm)	% Slow (Pool) Habitat
BCO_1	Above	41.3	7.2%	41.9	2.03	20.6	16%
	Below	0.4	4.7%	21.8	0.69	10.9	32%
BCT_1	Above	37.9	2.7%	50.7	1.78	19.6	0%
	Below	0.5	3.3%	42.5	1.39	13.5	47%
BOB_2	Above	73.2	3.3%	307.5	3.81	68.7	0%
	Below	2.9	4.6%	175	2.20	78.4	55%
CAN_1	Above	178.8	1.7%	117.3	3.68	24.8	39%
	Below	138.4	1.3%	114.2	3.52	5.1	5%
CAN_2	Above	36.8	1.7%	83.4	2.66	24.8	44%
	Below	10.7	1.3%	72.8	2.31	5.1	16%
CUR_1	Above	36.0	14.5%	95.7	2.31	124.0	15%
	Below	3.3	15.7%	58.4	1.34	54.3	58%
CUR_2	Above	5.8	14.5%	89.8	2.28	124.0	0%
	Below	2.1	15.7%	56.5	1.39	54.3	30%
FOX_1	Above	66.9	1.4%	159.0	4.56	11.8	15%
	Below	2.8	2.9%	69.7	1.92	41.6	44%
GRZ_1	Above	358.5	2.4%	364.4	5.87	116.0	0%
	Below	243.3	2.1%	285.8	4.56	91.9	13%
GRZ_2	Above	67.6	2.4%	251.5	4.10	116.0	4%
	Below	5.5	2.1%	145.5	2.36	91.9	60%
HAG_1	Above	332.4	2.9%	619.1	7.28	96.7	18%
	Below	138.0	3.3%	377.3	4.31	61.0	0%
HAG_2	Above	70.5	2.9%	535.3	6.30	96.7	11%
	Below	2.3	3.3%	220.9	2.58	61.0	40%
MIN_1	Above	101.6	10.4%	121.0	4.33	50.3	29%
	Below	38.1	10.0%	48.3	1.81	54.7	72%
MIN_2	Above	13.1	10.4%	106.2	3.91	50.3	43%
	Below	1.7	10.0%	45.5	1.63	54.7	70%
NFR_1	Above	368.9	3.7%	677.1	8.06	103.4	2%
	Below	0.3	3.1%	191.1	2.29	119.7	46%
NFR_2	Above	110.6	3.7%	571.5	6.85	103.4	9%
	Below	7.0	3.1%	219.7	2.56	119.7	17%
RAN_1	Above	180.9	10.6%	121.7	3.41	38.8	69%
	Below	138.2	7.1%	107.1	3.25	99.1	8%
SMN_2	Above	52.6	6.3%	327.7	3.67	99.3	0%
	Below	7.5	3.9%	249.1	2.78	77.5	27%
STL_1	Above	670.9	1.9%	402.0	6.34	75.9	2%
	Below	224.7	1.8%	358.8	5.64	76.1	3%
STL_2	Above	190.8	1.9%	365.9	5.80	75.9	0%
	Below	102.4	1.8%	337.6	5.30	76.1	0%

1 = summer visits and _2 = fall visits

APPENDIX B

PERCENT FINES MEASURED BY MULTIPLE TECHNIQUES

Table B.1 – Percent fines measured by multiple techniques

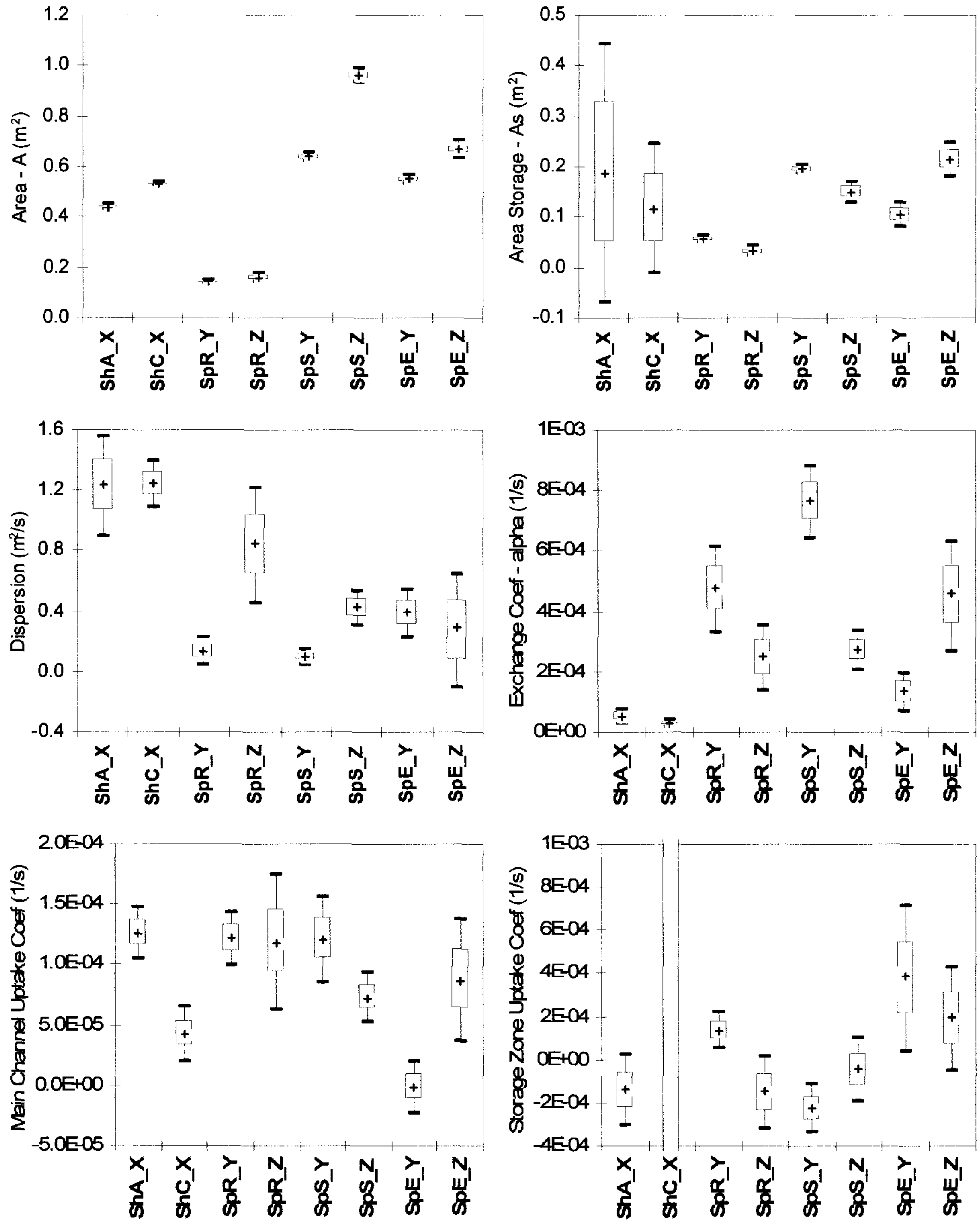
Site Code	Reach	Volumetric % Fines – Fast	Volumetric % Fines – Slow	Areal % Fines – Fast	Areal % Fines – Slow	Pebble Count % Fines	(Areal % Fines) - (Volumetric % Fines) Fast	(Areal % Fines) - (Volumetric % Fines) Slow
BCO_1	Above	2.0%	5.1%	8.7%	26.0%	12.0%	6.6%	20.9%
	Below	5.0%	6.7%	58.7%	85.3%	33.5%	53.7%	78.6%
BCT_1	Above	2.8%	3.5%	0.7%	38.0%	7.3%	-2.1%	34.5%
	Below	2.7%	2.4%	31.3%	69.3%	22.3%	28.7%	66.9%
BOB_2	Above	2.0%	4.5%	9.3%	32.0%	1.7%	7.3%	27.5%
	Below	1.6%	8.7%	3.3%	24.7%	6.0%	1.8%	16.0%
CAN_1	Above	3.6%	8.2%	2.0%	41.3%	11.3%	-1.6%	33.1%
	Below	7.9%	26.0%	34.7%	97.3%	18.0%	26.7%	71.4%
CAN_2	Above	4.5%	10.6%	1.3%	39.3%	11.3%	-3.1%	28.7%
	Below	7.2%	11.7%	5.3%	51.3%	18.0%	-1.9%	39.6%
CUR_1	Above	1.3%	1.2%	6.0%	15.3%	0.1%	4.7%	14.1%
	Below	2.1%	9.3%	22.7%	80.0%	6.6%	20.6%	70.7%
CUR_2	Above	0.5%	5.4%	3.3%	13.3%	0.1%	2.8%	7.9%
	Below	1.5%	6.5%	11.3%	25.3%	6.6%	9.9%	18.8%
FOX_1	Above	7.5%	12.7%	2.7%	37.3%	19.1%	-4.9%	24.6%
	Below	10.7%	7.3%	0.2%	0.7%	17.8%	-10.5%	-6.6%
GRZ_1	Above	1.1%	2.9%	2.0%	18.0%	1.5%	0.9%	15.1%
	Below	1.7%	3.9%	0.2%	8.7%	2.7%	-1.5%	4.8%
GRZ_2	Above	0.5%	3.2%	0.2%	6.7%	1.5%	-0.3%	3.5%
	Below	2.1%	3.5%	21.3%	9.3%	2.7%	19.2%	5.8%
HAG_1	Above	0.4%	2.2%	0.2%	0.1%	0.8%	-0.2%	-2.1%
	Below	1.0%	2.5%	1.3%	26.7%	3.8%	0.3%	24.2%
HAG_2	Above	0.9%	3.2%	0.7%	17.3%	0.8%	-0.2%	14.1%
	Below	1.6%	2.5%	2.0%	22.0%	3.8%	0.4%	19.5%
MIN_1	Above	0.6%	3.8%	2.0%	20.0%	11.0%	1.4%	16.2%
	Below	0.9%	0.8%	16.7%	76.0%	10.3%	15.7%	75.2%
MIN_2	Above	0.2%	8.1%	4.0%	51.3%	11.0%	3.8%	43.3%
	Below	1.5%	2.5%	20.0%	60.0%	10.3%	18.5%	57.5%
NFR_1	Above	4.0%	3.0%	5.3%	15.3%	0.1%	1.4%	12.3%
	Below	1.2%	1.2%	3.3%	8.7%	0.1%	2.2%	7.5%
NFR_2	Above	0.9%	3.7%	4.0%	16.0%	0.1%	3.1%	12.3%
	Below	0.3%	0.1%	0.2%	0.2%	0.1%	-0.2%	0.1%
RAN_1	Above	0.4%	8.3%	24.0%	68.0%	12.8%	23.6%	59.7%
	Below	0.8%	4.2%	10.0%	20.0%	3.3%	9.2%	15.8%
SMN_2	Above	0.9%	7.3%	4.0%	8.7%	1.0%	3.1%	1.3%
	Below	3.7%	2.3%	16.0%	56.0%	4.5%	12.3%	53.7%
STL_1	Above	1.1%	4.7%	0.2%	20.7%	1.2%	-0.9%	15.9%
	Below	2.0%	8.2%	5.3%	34.7%	4.4%	3.3%	26.4%
STL_2	Above	0.3%	5.3%	12.7%	5.3%	1.2%	12.4%	0.1%
	Below	0.9%	4.4%	3.3%	9.3%	4.4%	2.4%	4.9%
Mean		2.3%	5.5%	9.0%	31.4%	7.1%	16.2%[a]	52.6%[a]
Standard Deviation		2.4%	4.5%	12.0%	25.6%	7.6%		
Coefficient of Variation		1.042	0.812	1.336	0.817	1.060		

_1 = summer visits and _2 = fall visits

[a] *Absolute average (average of absolute values)*

APPENDIX C

GRAPHICAL COMPARISONS OF INDIVIDUAL OTIS-UCODE PARAMETERS



Note: Box plots display 10th, 25th, 50th, 75th, and 90th percentiles and standard deviations for the variable A of SpR_Z, SpS_Y, and SpS_Z were set to averaged values across the remaining five injections.

Figure C.1 – Monte Carlo derived distributions for transient storage and nutrient uptake variables

APPENDIX D

SUMMARY OF GEOMORPHIC COMPLEXITY METRICS

Table D.1 – Summary of geomorphic complexity metrics

Visit Code	ShA_X	ShB_X	ShC_X	SpE_X	SpE_Y	SpE_Z	SpR_X	SpR_Y	SpR_Z	SpS_X	SpS_Y	SpS_Z
Study Date	7/15/07	7/16/07	7/17/07	6/25/07	7/30/07	8/6/07	6/28/07	8/1/07	8/8/07	6/26/07	7/31/07	8/9/07
Flow Rate (L/s)	108.0	88.0	102.0	156.5	72.0	152.0	70.0	16.5	20.6	133.0	46.0	108.0
Water Surface Slope	1.46%	0.70%	1.15%	0.44%	0.44%	0.44%	0.24%	0.24%	0.24%	0.64%	0.64%	0.64%
Reach Length (m)	184.1	192.2	190.6	177.9	177.9	177.9	179.7	179.7	179.7	180.0	180.0	180.0
Channel Sinuosity	1.084	1.905	1.239	1.159	1.159	1.159	1.007	1.007	1.007	1.048	1.048	1.048
Number of Habitat Units	23	28	18	12	12	12	1	1	1	10	10	10
Surveyed Planform Area (m^2)	559.3	537.6	828.8	820.5	820.5	820.5	576.2	576.2	576.2	820.5	820.5	820.5
Metric of Complexity X[a] (m)	0.00129	0.00135	0.00162	0.00081	0.00081	0.00081	0.00015	0.00015	0.00015	0.00114	0.00114	0.00114
Average Water Surface Concavity AWSC[b]	0.00309	0.00119	0.00329	0.00063	0.00063	0.00063	0.00094	0.00094	0.00094	0.00360	0.00360	0.00360
Average Thalweg Concavity	0.00850	0.00807	0.03374	0.00449	0.00449	0.00449	0.00179	0.00179	0.00179	0.02646	0.02646	0.02646
Longitudinal Roughness	0.59	0.77	0.99	1.04	1.42	0.99	1.08	0.63	0.72	0.91	1.12	0.77
Thalweg Variation	5428.4	3816.0	3781.8	1741.5	2385.8	1664.6	2090.1	1221.3	1383.5	4582.7	5652.3	3903.1
Width Residual	0.18	0.23	0.23	0.28	0.31	0.31	0.18	0.27	0.28	0.20	0.21	0.22
Width Variation	1082.5	2060.1	897.2	541.3	614.4	611.3	538.4	819.1	855.3	907.7	980.2	993.6

[a] Gooseff *et al.* (2007)

[b] Anderson *et al.* (2005)

APPENDIX E

GEOMORPHIC COMPLEXITY PEARSON CORRELATION MATRIX

Table E.1 – Geomorphic complexity Pearson correlation matrix

Variable	So	f	Sin	Num_U	LR	TV	WV	WR	GM	AWSC	AThC	D_LR	D_TV	D_WV	D_WR	fines	d16	d50	d84	s_grad	s_stdv	s_fred	s_cv
So	1.00	0.10	0.43	0.92	-0.03	0.60	0.68	-0.06	0.83	0.69	0.48	-0.17	0.76	0.61	-0.75	-0.57	0.07	0.67	0.82	0.71	0.74	0.71	-0.50
f	0.10	1.00	-0.24	0.05	0.57	0.78	0.70	0.07	0.39	0.69	0.65	0.23	0.70	0.48	-0.53	-0.18	0.22	0.11	0.20	-0.21	-0.12	0.21	-0.33
Sin	0.43	-0.24	1.00	0.65	0.37	-0.05	0.30	0.75	0.64	0.05	0.34	0.55	0.09	-0.36	0.15	-0.54	0.23	0.58	0.53	0.47	0.50	0.49	-0.54
Num_U	0.92	0.05	0.65	1.00	0.26	0.54	0.63	0.31	0.87	0.51	0.39	0.13	0.68	0.29	-0.49	-0.72	0.30	0.84	0.89	0.56	0.63	0.81	-0.70
LR	-0.03	0.57	0.37	0.26	1.00	0.49	0.51	0.81	0.48	0.25	0.45	0.70	0.32	-0.23	0.06	-0.56	0.69	0.51	0.42	-0.36	-0.22	0.50	-0.67
TV	0.60	0.78	-0.05	0.54	0.49	1.00	0.90	0.01	0.71	0.89	0.68	-0.03	0.92	0.72	-0.82	-0.56	0.40	0.56	0.69	0.08	0.19	0.68	-0.58
WV	0.68	0.70	0.30	0.63	0.51	0.90	1.00	0.20	0.90	0.95	0.91	0.12	0.88	0.63	-0.78	-0.63	0.34	0.61	0.76	0.33	0.42	0.73	-0.64
WR	-0.06	0.07	0.75	0.31	0.81	0.01	0.20	1.00	0.40	-0.12	0.24	0.84	-0.04	-0.63	0.44	-0.50	0.54	0.47	0.33	-0.13	-0.04	0.38	-0.58
GM	0.83	0.39	0.64	0.87	0.48	0.71	0.90	0.40	1.00	0.77	0.79	0.25	0.78	0.40	-0.61	-0.75	0.35	0.78	0.89	0.52	0.60	0.83	-0.74
AWSC	0.69	0.69	0.05	0.51	0.25	0.89	0.95	-0.12	0.77	1.00	0.86	-0.15	0.89	0.84	-0.92	-0.46	0.15	0.44	0.65	0.38	0.43	0.59	-0.44
AWThC	0.48	0.65	0.34	0.39	0.45	0.68	0.91	0.24	0.79	0.86	1.00	0.17	0.66	0.50	-0.63	-0.46	0.20	0.39	0.56	0.36	0.41	0.52	-0.47
D_LR	-0.17	0.23	0.55	0.13	0.70	-0.03	0.12	0.84	0.25	-0.15	0.17	1.00	0.04	-0.57	0.43	-0.21	0.11	0.09	-0.01	-0.05	0.03	0.00	-0.45
D_TV	0.76	0.70	0.09	0.68	0.32	0.92	0.88	-0.04	0.78	0.89	0.66	0.04	1.00	0.74	-0.86	-0.49	0.15	0.52	0.67	0.35	0.44	0.60	-0.57
D_WV	0.61	0.48	-0.36	0.29	-0.23	0.72	0.63	-0.63	0.40	0.84	0.50	-0.57	0.74	1.00	-0.96	-0.15	-0.14	0.15	0.38	0.38	0.39	0.31	-0.09
D_WR	-0.75	-0.53	0.15	-0.49	0.06	-0.82	-0.78	0.44	-0.61	-0.92	-0.63	0.43	-0.86	-0.96	1.00	0.28	0.01	-0.34	-0.56	-0.41	-0.43	-0.49	0.23
fines	-0.57	-0.18	-0.54	-0.72	-0.56	-0.56	-0.63	-0.50	-0.75	-0.46	-0.46	-0.21	-0.49	-0.15	0.28	1.00	-0.68	-0.89	-0.87	-0.27	-0.43	-0.90	0.94
d16	0.07	0.22	0.23	0.30	0.69	0.40	0.34	0.54	0.35	0.15	0.20	0.11	0.15	-0.14	0.01	-0.68	1.00	0.76	0.62	-0.46	-0.34	0.73	-0.57
d50	0.67	0.11	0.58	0.84	0.51	0.56	0.61	0.47	0.78	0.44	0.39	0.09	0.52	0.15	-0.34	-0.89	0.76	1.00	0.96	0.17	0.28	0.97	-0.79
d84	0.82	0.20	0.53	0.89	0.42	0.69	0.76	0.33	0.89	0.65	0.56	-0.01	0.67	0.38	-0.56	-0.87	0.62	0.96	1.00	0.34	0.44	0.99	-0.76
s_grad	0.71	-0.21	0.47	0.56	-0.36	0.08	0.33	-0.13	0.52	0.38	0.36	-0.05	0.35	0.38	-0.41	-0.27	-0.46	0.17	0.34	1.00	0.98	0.21	-0.27
s_stdv	0.74	-0.12	0.50	0.63	-0.22	0.19	0.42	-0.04	0.60	0.43	0.41	0.03	0.44	0.39	-0.43	-0.43	-0.34	0.28	0.44	0.98	1.00	0.32	-0.44
s_fred	0.71	0.21	0.49	0.81	0.50	0.68	0.73	0.38	0.83	0.59	0.52	0.00	0.60	0.31	-0.49	-0.90	0.73	0.97	0.99	0.21	0.32	1.00	-0.77
s_cv	-0.50	-0.33	-0.54	-0.70	-0.67	-0.58	-0.64	-0.58	-0.74	-0.44	-0.47	-0.45	-0.57	-0.09	0.23	0.94	-0.57	-0.79	-0.76	-0.27	-0.44	-0.77	1.00

APPENDIX F

TRANSIENT STORAGE AND NUTRIENT UPTAKE METRICS PARAMETERIZED BY UCODE/OTIS MODELING

Table F.1 – Transient storage and nutrient uptake metrics parameterized by UCODE/OTIS modeling

Injection Code	Var	Optimal Value (μ)	Standard Deviation (σ)	Coefficient of Variation (CV)	Comp. Scaled Sensitivity (CSS)	Ratio to Maximum CSS[a]
ShA_X	A	0.44	6.7E-03	0.02	6.60	1.000
	A_s	0.19	2.1E-01	1.12	0.08	0.013
	D	1.23	2.4E-01	0.20	0.47	0.071
	α	5.2E-05	1.9E-05	0.37	0.27	0.041
	λ	1.3E-04	1.7E-05	0.13	4.99	1.000
	λ_s	-1.4E-04	1.2E-04	-0.88	0.68	0.135
ShC_X	A	0.53	5.4E-03	0.01	6.10	1.000
	A_s	0.12	1.0E-01	0.89	0.06	0.010
	D	1.25	1.1E-01	0.09	0.64	0.104
	α	2.8E-05	9.6E-06	0.34	0.19	0.031
	λ	4.3E-05	1.8E-05	0.42	6.79	1.000
	λ_s	6.0E-02	1.5E+01	255.36	0.0005	0.0001
SpR_Y	A	0.14	4.3E-03	0.03	10.04	1.000
	A_s	0.06	4.9E-03	0.09	1.62	0.162
	D	0.13	6.5E-02	0.49	0.30	0.030
	α	4.8E-04	1.1E-04	0.23	1.28	0.127
	λ	1.2E-04	1.8E-05	0.15	8.96	1.000
	λ_s	1.4E-04	6.2E-05	0.46	2.58	0.288
SpR_Z	A	0.16[b]				
	A_s	0.03	8.3E-03	0.26	0.76	1.000
	D	0.84	2.9E-01	0.34	0.58	0.765
	α	2.5E-04	8.5E-05	0.34	0.54	0.706
	λ	1.2E-04	4.4E-05	0.37	6.13	1.000
	λ_s	-1.5E-04	1.3E-04	-0.84	2.16	0.352
SpS_Y	A	0.64[b]				
	A_s	0.19	7.5E-03	0.04	3.06	1.000
	D	0.10	3.6E-02	0.37	0.40	0.132
	α	7.7E-04	9.3E-05	0.12	1.17	0.383
	λ	1.2E-04	2.8E-05	0.23	17.39	1.000
	λ_s	-2.2E-04	8.2E-05	-0.37	5.97	0.343
SpS_Z	A	0.96[b]				
	A_s	0.15	1.8E-02	0.12	0.81	1.000
	D	0.42	8.4E-02	0.20	0.49	0.608
	α	2.7E-04	5.1E-05	0.19	0.48	0.594
	λ	7.3E-05	1.7E-05	0.23	8.98	1.000
	λ_s	-4.3E-05	1.1E-04	-2.58	1.35	0.150
SpE_Y	A	0.55	1.5E-02	0.03	8.36	1.000
	A_s	0.11	2.0E-02	0.19	0.54	0.065
	D	0.39	1.2E-01	0.31	0.43	0.052
	α	1.3E-04	5.0E-05	0.38	0.59	0.070
	λ	-1.1E-06	1.7E-05	-15.52	0.29	1.000
	λ_s	3.8E-04	2.5E-04	0.65	0.02	0.067
SpE_Z	A	0.67	2.5E-02	0.04	6.11	1.000
	A_s	0.21	2.8E-02	0.13	0.95	0.155
	D	0.27	2.8E-01	1.03	0.13	0.022
	α	4.6E-04	1.4E-04	0.31	0.70	0.114
	λ	8.8E-05	4.1E-05	0.46	0.18	1.000
	λ_s	2.0E-04	1.8E-04	0.92	0.04	0.227

[a] Transient storage model and nutrient uptake models were ran separately; maximum CSS reference is either to A, A_s, D, and α, of each site, or λ and λ_s.

[b] To obtain acceptable model for this injection, area was fixed thereby precluding uncertainty analysis.

Table F.2 – Correlation matrices for transient storage variables

Injection	Variable	A	A_s	D	α
ShA_X	A	1.00	0.39	0.32	-0.44
	As	0.39	1.00	0.07	-0.29
	D	0.32	0.07	1.00	-0.37
	α	-0.44	-0.29	-0.37	1.00
ShC_X	A	1.00	0.38	0.20	-0.43
	As	0.38	1.00	0.09	-0.19
	D	0.20	0.09	1.00	-0.41
	α	-0.43	-0.19	-0.41	1.00
SpR_Y	A	1.00	-0.58	0.61	-0.92
	As	-0.58	1.00	-0.51	0.53
	D	0.61	-0.51	1.00	-0.58
	α	-0.92	0.53	-0.58	1.00
SpR_Z	A[a]				
	As		1.00	-0.42	-0.09
	D		-0.42	1.00	0.07
	α		-0.09	0.07	1.00
SpS_Y	A[a]				
	As		1.00	-0.50	-0.40
	D		-0.50	1.00	0.70
	α		-0.40	0.70	1.00
SpS_Z	A[a]				
	As		1.00	-0.41	-0.11
	D		-0.41	1.00	0.22
	α		-0.11	0.22	1.00
SpE_Y	A	1.00	-0.17	0.62	-0.89
	As	-0.17	1.00	-0.34	0.16
	D	0.62	-0.34	1.00	-0.62
	α	-0.89	0.16	-0.62	1.00
SpE_Z	A	1.00	-0.50	0.65	-0.88
	As	-0.50	1.00	-0.50	0.38
	D	0.65	-0.50	1.00	-0.61
	α	-0.88	0.38	-0.61	1.00

* Area (A) had to be set to a fixed value in this model, therefore, no estimate for correlation is available.

KEY:
R > |0.95|
R > |0.90|
R > |0.85|
R ≤ |0.85|

Table F.3 – Correlation matrices for nutrient uptake variables

Injection	Variable	λ	λ_s
ShA_X	λ	1.00	-0.12
	λ_s	-0.12	1.00
ShC_X	λ	1.00	-0.94
	λ_s	-0.94	1.00
SpR_Y	λ	1.00	-0.79
	λ_s	-0.79	1.00
SpR_Z	λ	1.00	-0.55
	λ_s	-0.55	1.00
SpS_Y	λ	1.00	-0.93
	λ_s	-0.93	1.00
SpS_Z	λ	1.00	-0.85
	λ_s	-0.85	1.00
SpE_Y	λ	1.00	-0.67
	λ_s	-0.67	1.00
SpE_Z	λ	1.00	-0.78
	λ_s	-0.78	1.00

KEY:
R > |0.95|
R > |0.90|
R > |0.85|
R ≤ |0.85|