Channel Morphology and Riparian Vegetation Influences on Fluvial Aquatic Habitat

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Abstract

As public awareness of river degradation has grown in recent years, the number of stream restoration activities has increased dramatically. Anthropogenic influences at a range of spatial scales from watershed landuse to riparian vegetation management to local channel morphology can have hierarchal relationships to local (meso- and macro-) in-stream habitat characteristics. This research examined these influences first by examining the influence of complex channel morphology on meso-scale brook trout (*Salvelinus fontinalis*) habitat in Shenandoah National Park, VA, and then by examining the combined influence of watershed urbanization and riparian vegetation (100-200 m reaches) on stream temperature.

Moving beyond one-dimensional (1D) averaged representations of fish habitat, this research explored the distribution of two-dimensional (2D) flow complexity metrics at the meso-habitat scale as explanatory variables for brook trout habitat preferences and as potential metrics to evaluate habitat restoration design. Spatial hydraulic complexity metrics, including area-weighted *circulation* and *kinetic energy gradients,* were calculated based on 2D depth averaged modeled velocity distributions in two 100-m reaches on the Staunton River. While there were no statistically significant correlations between *kinetic energy gradients* or area-weighted *circulation* and fish density, fish density was positively correlated to the percent of the channel dominated by protruding boulders. The structural complexity of areas with protruding boulders create complex flow patterns suggesting that flow complexity plays an important role in available brook trout habitat preferences at the local scale, although the 2D depth averaged model may not have adequately represented this complexity. The 2D distribution of flow characteristics was then investigated further to quantify areas of flow refugia (low velocity shelters) and the relationship between these areas, traditional measures of habitat quality, and fish biomass. Flow complexity in the vicinity of flow obstructions (in this case, boulders) was investigated further using patch classification and landscape ecology metrics.

The relative influence of riparian vegetation on stream temperature (another important habitat characteristic) in urban and nonurban watersheds was investigated in 27 paired forested and nonforested reaches in PA, MD, and DE. Riparian vegetation and watershed-scale

urbanization both influence stream temperature, which can have profound impacts on in-stream ecosystems. Generally, increased urbanization and removal of riparian forest influenced maximum stream temperatures resulting in higher maximum summer stream temperatures (up to 1.8 ^{o}C); however, the influence of riparian forests (at at 100-200 m reach scale) decreased with increasing urbanization. Extreme maximum summer temperatures, which are a concern for aquatic biota, increased in both frequency and duration in urban nonforested reaches relative to forested reaches indicating that the addition of a forested 100-200 m long buffer partially mitigated these temperature extremes even in urban watersheds. Overall, changes to channel morphology and riparian vegetation had measureable local effects on stream habitat (temperature and hydraulic complexity) yet the implications of restoration efforts at the local scale on ecosystem services at a larger (km +) scale requires further study.

Dedication

I dedicate this work in memory of Julia Pryde whose laughter infected all who knew her.

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Introduction and Problem Description

As public awareness of river degradation has grown in recent years, the number of stream restoration activities has increased dramatically. Leading causes of stream impairment include pathogens, siltation, habitat alteration, oxygen-depleting substances, nutrients, and thermal enhancement (U.S. EPA 2000). While these impairments result from multiple sources including point sources, landuse, nonpoint source (NPS) pollution, and stream channel modification, the terms stream restoration or rehabilitation typically address those impairments such as habitat alteration, siltation, and thermal enhancement, that are directly influenced by channel or riparian corridor management (Shields et al. 2003). Habitat alteration in this case refers to alterations to the physical structure of the stream channel or riparian area (e.g. removal of woody debris), and not to stream flow alterations (e.g. dams). Stream restoration refers to a range of management activities in the stream channel or riparian corridor with goals of water-quality improvement, bank stabilization, riparian management, fish passage, and in-stream habitat improvement.

Despite large expenditures (greater than 1 billion dollars annually since 2000) there is still little information available about the motivation, actions, and results of stream or river restoration projects (Bernhardt et al. 2007). While the success of these projects is dependent on the specific project goals and cannot be evaluated without long-term monitoring, guidance available for the engineering design of in-stream habitat tends to be qualitative, and the implications of restoration activities on specific habitat characteristics such as in-stream temperature is still largely unknown. Historically, the objective of river engineering was to remove obstructions to flow to speed the passage of floods; however, as the understanding of riverine systems has improved, many restoration projects aim to restore some level of complexity to chanelized streams, or to reforest nonforested riparian areas.

There are many factors affecting local fish communities including temperature, cover, substrate, water quality, food supply, predation. This research is specifically geared toward stream restoration projects where in-stream fish habitat improvement is a restoration goal and focuses on two aspects of stream restoration relevant to stream habitat, the relationship between channel morphology and hydraulic habitat, and riparian vegetation and stream temperature.

Research Objectives

The overall goal of this research was to assess the characteristics of in-stream habitat that need to be considered for successful stream habitat restoration. Specifically, this research aimed to explore and define the relationships between channel morphology, riparian vegetation, and watershed imperviousness on in-stream habitat.

Specific objectives include the following:

1. Evaluate various flow complexity and local habitat metrics to determine their explanatory relationship to fish habitat preference.
2. Explore the spatial distribution of flow and habitat variables across a range of flows.
3. Quantify the relative influence of riparian vegetation on stream temperature characteristics in urban and nonurban streams.

Chapter 1: Literature Review

1.1. Instream Habitat

In-stream habitat refers to the physical habitat, or "living", space of in-stream biota and encompasses the channel's physical structure, flow regime, and their spatial and temporal dynamics (Maddock, 1999). The most important reach scale abiotic factors that affect fish in running water are temperature (directly and indirectly through oxygen consumption), rate and fluctuation of discharge, and availability of suitable shelter (substratum) (Hynes, 1970); however, microhabitat characteristics including depth, velocity, substrate, and cover are biologically important to fish for spawning, feeding, and refugia during high flows (Maddock, 1999; Rempel et al., 1999; Schwartz and Herricks, 2005; Smith et al., 2006). Understanding the linkages between physical, chemical, biological components of the river environment is essential to restore and maintain ecological function (Kemp et al. 2000).

1.1.1. Stream Temperature and Aquatic Habitat

Temperature is one of the most important reach-scale abiotic factors affecting fish in running water (Hynes 1970). Of particular importance are extreme high temperatures which can increase biological rates resulting in decreased dissolved oxygen, limited fish habitat (based on species and life stage thermal preferences), and ultimately increase fish mortality (see review by Caissie 2006). At these elevated temperatures, thermal habitat complexity becomes important (thermal refugia) and the timing and duration of temperature events can increase stress. Increased stream temperatures can shift species communities away from intolerant species such as macroinvertebrate taxa (i.e. stoneflies) that are more thermally sensitive to elevated temperature adversely affecting biological diversity (Krause et al. 2004).

Poole and Berman (Poole and Berman 2001) describe stream temperature as a function of heat load divided by discharge, where heat load is a measure of heat energy added to a stream; therefore, stream temperature is affected by heat load dynamics at a range of temporal scales (i.e. daily fluctuations in solar energy, and annual fluctuations in air temperature) in addition to the temporal dynamics of the flow regime.

Anthropogenic stressors associated with urban land use, such as increased watershed imperviousness, destruction of riparian vegetation, and increased siltation, are expected to alter

stream temperature regimes (Nelson and Palmer 2007). The increase in impervious surfaces associated with urbanization affects both stream discharge and heat load. Increased imperviousness leads to a number of hydrologic changes including reduced infiltration, which in turn can reduce baseflow and increase the flashiness of urban streams. The runoff over heated parking lots and other imperious surfaces is a source of heat load to urban streams (Van Buren et al. 2000; Herb et al. 2008). Generally, increased urbanization has been found to result in higher stream temperatures (LeBlanc et al. 1997; Poole and Berman 2001; Krause et al. 2004). Elevated stream temperature in urban streams is commonly attributed to changes to riparian vegetation, stream width, streamflow, and groundwater (Krause et al. 2004). Temperature surges caused by thermal enhancement of stormwater runoff over impervious surfaces are also an important mechanism for thermal pollution in urban areas (Van Buren et al. 2000; Nelson and Palmer 2007). Other potential factors include the removal or introduction of wetlands or ponds, and point source thermal discharges (LeBlanc et al. 1997).

1.1.2. Habitat and Scale

Habitat restoration can occur at a range of scales from microscale (e.g. individual habitat structures such as boulders) to macrohabitat (e.g. pool-riffle sequences) to reach (e.g. creation of meanders) to stream-system (e.g. removal of man made obstructions, dams) (Maddock 1999). Based on the idea that species assemblages and/or abundances are organized to reflect physical conditions at a range of scales cascading from watershed to microhabitat, watershed scale conditions such as slope, geology, and precipitation determine the type of species that may be present, in the reach, subreach, or individual habitat patches (Clifford et al. 2008). Richards et al. (1996) examined the relative influence of geologic versus anthropogenic catchments attributes on stream ecosystems in 45 catchments of a single river basin in central Michigan. While stream buffers (100 m) were more important for predicting sediment-related variables, channel morphology was more strongly related to catchment-scale characteristics (i.e. land use, geology, elevation and hydrography) indicating that for physical habitat, understanding catchment scale characteristics is essential for restoring stream ecosystems. Bond and Lake (2003) suggested that, while fish respond to habitat at several spatial scales, in systems where fish abundances are limited by low habitat availability at small (meter) spatial scales, fish abundances could be increased by augmenting the amount of available habitat via stream restoration, although

numerous studies fail to detect the effects of local habitat restoration (e.g. Roni et al. 2002 and Lepori et al. 2005). Channel form, instream habitat, and stream communities appear to be hierarchically related, but the strength of the relationships among all components of this hypothesized hierarchy is not fully understood. In a study by Smiley and Dibble (2005), fish and macroinvertebrate communities generally exhibited stronger relationships with instream habitat (depth, velocity, and substrate type) than with channel form. Species richness, evenness, and abundance tended to exhibit greater correlations with instream habitat, while species composition had greater correlations with channel form. Other studies have suggested that in addition to subunit (patch) area, the habitat heterogeneity determined within habitat units (riffles, pools, etc.) in an hierarchical framework may be more relevant to fish populations (Thomson et al. 2001).

At the reach and subreach scale, interactions between the physical and ecological environments are driven by hydraulics. Distinction is made between habitat units or biotopes (channel features characterized through hydraulic measurement or visual survey of surface flow type; e.g. Kemp et al. 2000; Newson and Newson 2000) and functional habitats (discrete species assemblages associated with meso-habitats) (Clifford et al. 2008). Surveyed habitat units are intended to represent integrated habitat as a means to avoid issues relating to single-species preference curve or single measures of channel environment; however, the relationships between habitat units and ecological function are uncertain (Kemp et al. 2000), in part due to the difficulty in defining representative units across a range of flows and species life stage (Clifford et al. 2008). When comparing habitat units (e.g. pools and riffles) and subunits (patches within channel units defined by depth, velocity and substrate) to salmon abundance, Inoue and Nunokawa (2002) found that the longitudinal fish abundance variation was related to the subunit scale (pool heads) habitat patches.

Ecological observations that species patchiness occurs in a general downstream continuum supports the general assumption that physical habitat units (e.g. riffles and pools) describe biologically relevant habitat areas (Clifford et al. 2008), yet the relationships between habitat units and biological function are poorly understood (Kemp et al. 2000). Descriptions of habitat areas utilized by individual fish such as brook trout fish selecting deep, low current velocity locations (Sotiropoulos et al. 2006) indicate that physical habitat areas can be discriminated based on hydraulic characteristics. One such combined index that has been utilized to define habitat patches is the Froude (Fr) number (ratio of velocity to the square root of

depth multiplied by gravitational acceleration, or the ratio of inertial to gravitational forces). Kemp et al. (2000) suggest that functional habitats can be divided into groups based on greater than or less than Fr = 0.5, while (Jowett 1993) had three ranges identifying pool, riffle, and run habitat. However, using Fr to identify habitats can obscure differences between very different flow-depth combinations.

Often rapid habitat assessment is conducted based on a visual classification surface flow type (Thomson et al. 2001). Newson et al. (1998) identified 10 different physical habitat units (e.g. riffle, pool, cascade) based on descriptions of surface flow each of which has been proved as distinctive by hydraulic study. Rabeni and Jacobson (1993) classified hydraulic habitat units in terms of three different kinds of pools (scour, slack water, and transistion) which were further subdivided using descriptive characteristics such as location or typical cover in each habitat unit. In this study, pools formed by flow obstructions comprised less than 1% of total area, but were expected to be important biologically because of its complex mixture of depths, velocity and physical structure.

Visual methods of habitat unit classification can vary by user and are largely dependent on flow condition at the time of the survey. Less subjective methods of habitat unit or patch classification demand relatively high-resolution spatially distributed values of hydraulic characteristics. Emery et al. (2003) classified habitat patches using hierarchical cluster analysis (Ward's algorithm) as a reproducible means of accessing the degree, location and persistence of patchiness of physical habitat. The data requirements for clustering methods suggest the need for at least 2Dimentional (2D) model results or remote sensing of channel morphology and flow characteristics or a combination of methods (Legleiter and Goodchild 2005; Marchamalo et al. 2007). While classifying the area of individual habitat patches may be useful for describing habitat availability, the heterogeneity of patches determined within habitat units (riffles, pools, etc.) in an hierarchical framework may be more relevant to fish populations (Thomson et al. 2001).

1.1.3. Spatial Complexity

In many ecological systems, biotic diversity is positively correlated with habitat heterogeneity (Bell et al., 1991; Rosenzweig, 1995). Habitat heterogeneity in streams has been shown to be related to macroinvertebrate taxon richness, fish species diversity and density,

periphyton, and processing of coarse particulate organic matter (Gorman and Karr, 1978; Biggs and Stokseth, 1996; Brown, 2003; Lepori et al., 2005a).

Often restoration measures aim to add a level of complexity to channelized streams by introducing flow obstructions in various configurations with varying effects on fish populations. Rhoads et al. (2003) found that a channelized reach had significantly less species richness, species diversity, and total biomass than an unmodified reach immediately upstream, suggesting that three-dimensional (3D) habitat hydraulics driven by structural complexity appears to strongly influence fish community characteristics. This was supported by Shields et al. (1995) who found that fish lengths increased with the introduction of restoration structures (weirs). Lepori et al. (2005) found little effect of restoration schemes on biodiversity indicating that in this case the structural heterogeneity was not relevant to the target species. In addition, the Dolinsek et al. (2007) found that the addition of boulders increased the density of salmonid fishes by 2.8-fold, but had no significant effect on non-territorial fishes and that the effect of increased habitat complexity was dependent on fish life stage. While fish density often increases with structural complexity due in part to the protection offered from predators and aggressive competitors, the relationships between fish populations and habitat complexity is confounded by interactions between behavioral variation and territory size (Hojesjo et al. 2004).

1.2. Habitat Modeling

Current literature indicates that there is a disconnect between fish habitat preferences at the local (micro) scale and predictive measures of habitat quality at meso- and macro- (reach or larger) scale. Often, local habitat characteristics such as velocity, depth and substrate at individual fish locations are combined into preference curves which are then used to scale-up local preferences to meso- or reach scale habitat quality. There are several hydraulic models that have been used to simulate hydraulic habitat suitability. The Physical Habitat Simulation Model (PHABSIM) is a well-established hydro-ecological model that provides a suite of tools for the numerical modeling of hydraulic habitat suitability for fish and invertebrate species based on field measurements of channel slope, water depth, velocity, and substrate (Bovee, 1982; Maddock, 1999; Booker and Dunbar, 2004); however, PHABSIM is one-dimensional (1D) and averages velocity, substrate, and depth values between measured cross sections and cannot account for the range of habitat types or physical conditions adjacent to a fish location within a

stream (Maddock, 1999).   The strengths of this methodology are its simplicity and ability to evaluate a range of flows (Clifford et al. 2008), but results are limited by the applicability of habitat suitability indices (HSIs) for target species (Booker et al. 2004).

Clifford et al. (2008) categorized HSIs into four groups: Type 1 HSIs are derived from literature or expert opinion; Type 2 are derived through frequency analysis of physical habitat utilized by different species or life stages in field observations; Type 3 HSIs (preference curves) have been corrected for habitat availability; and, Type 4 are multi-variate HSIs that weight habitat suitability based on depth and velocity together.   Alternatives to using HSIs include evaluating physical habitat based on maximum swimming speed for target species based on the theory that excessive velocities at high flows are one factor that limits fish habitat (Booker 2003) and bioenergetic models (Booker and Dunbar 2004; Hayes et al. 2007) that compare the energetic gain from foraging to the energy expenditure in different habitat areas. While the use of HSI indices are a major source of uncertainty in PHABSIM modeling (Moir et al. 2005), the use of this modeling scheme is more limited by inability to model spatial variation in flow characteristics.

Two- and three-dimensional (2D and 3D) hydraulic models have been used to calculate flow characteristics in streams as a measure of habitat suitability (Booker, 2003; Clark et al., 2006; Crowder and Diplas, 2006). The advantage of 2D and 3D habitat models over the conventional 1D model PHABSIM is the ability to spatially determine depth and depth-averaged velocity and flow direction. This allows the user to evaluate areas of particular ecological importance such as refugia from high flows.

Similar to PHABSIM, a 2D hydraulic model, River2D, can be combined with habitat suitability indices (HSI) for target species to calculate the weighted usable area (WUA), or relative amount of preferred habitat available to a target species (Ghanem et al., 1996; Steffler and Blackburn, 2002). However, WUA ignores local flow complexity and predicts habitat suitability based on species flow and depth requirements while the spatial distribution of WUA or habitat units may provide a more realistic view of habitat within the stream. Previous research using River2D to examine the distribution of WUA found that reach-averaged WUA did not accurately predict fish populations and that the distribution of usable habitats was a better determining factor for fish populations. Clark et al. (2008) found that streams with discontinuous, distinct patches of high score WUA had lower fish biotic integrity than streams

with a more uniform distribution of high WUA. Instead of using WUA, Stewart et al. (2005) classified habitat units based on expert observations and calculated the heterogeneity of habitats by the spatial diversity of different habitat units. This 2D habitat heterogeneity was positively correlated to fish biomass indicating that the spatial structure of habitat units is also important when evaluating habitat. This spatial structure cannot be captured in a 1D model.

1.3. *Hydraulic Modeling*

Models of flow complexity need to capture secondary currents such as the large wake areas behind boulders. Flow obstructions, or obstacles to the mean motion of flow, create variations in flow patterns in rivers and streams. The flow around, over, and through obstructions creates complex three-dimensional (3D) flow patterns. These obstructions include protruding elements (such as large boulders and large woody debris) and submerged elements (boulders, gravel clusters, emergent vegetation). The flow variability created by these obstructions can create areas of refugia during high flows, preferential transport of materials, can encourage mixing, and can provide slower moving areas of flow for the breakdown/storage of organic matter or chemical contaminants. This variability is important because different in-stream species can tolerate different ranges of flow velocities, water depths, bed substrates, and water quality. Hydraulic habitat heterogeneity, defined by secondary currents, turbulence, and variation in local flow velocity and depth, encourages the diversity of river benthos and fish species (Bockelmann et al. 2004).

One of the simplest cases of flow around an obstruction is flow around a vertical cylinder such as a bridge pier. In the wake region behind the cylinder, vortices are formed and shed periodically from alternating sides the cylinder (Crowe, et al., 2002). This vortex street is characteristic of the flow behind a cylindrical obstruction. In addition, a flow visualization test conducted by Sadeque et al. (2008) confirmed the formation of a horse-shoe vortex system at the point of flow separation upstream of a protruding cylinder.

When flow obstructions are submerged, flow variability and turbulent structures become more complex. When Sadeque et al. (2008) conducted flow visualizations of moderately to deeply submerged cylinders in flow a closed wake downstream of the obstruction was distinguishable attributed to faster moving shear layers creating a 3D envelope behind the shallow objects. This accelerated flow interfered with the formation of wake vortices. However,

for cylinders that were only slightly submerged, the vortex street was a characteristic feature of the flow.

Similar to the simplified case of cylinders, boulders protruding from the flow have been shown to produce coherent wake structures with vertical axes of rotation. The kinetic energy of the vertical vortex structures can be utilized by swimming fish (Liao et al., 2003). However, the complex geometry of natural boulders makes it difficult to predict the vortex shedding frequency (Tritico and Hotchkiss, 2005).

Flow around and over submerged boulders produces spatial heterogeneity of near-bed flows (Shen and Diplas 2008). Consistent with the Sadeque et al. (2008) study, this response is attributed to the interaction of the horseshoe vortices developed immediately behind the boulder with strong arch-type vortices shed from the top of the boulders. This influence decreases with height above the boulder in deep flows as the flow patterns become more uniform. The 3D flow around and over top of the submerged boulder creates areas of high vorticity behind boulders.

Strom et al (2007) conducted a flume study evaluating flow variability around self-formed bed clusters. The log-law type velocity relation held for regions outside of the influence of a cluster microform; however, the logarithmic velocity distribution breaks down in the near-wall region around the cluster. Similar to the flow over submerged cylinders, this study was unable to detect a well-defined horseshoe vortex associated with the cluster. And similar to flow over submerged boulders, the effects due to the presence of the cluster were not felt about 0.2 times the height of the cluster from the bed. There was a general upwelling over the cluster stoss and downwelling in the wake. Embedded within this larger scale were smaller recirculation pockets attributed to flow around individual particles. This heterogeneity leads to pockets of high and low friction velocity and pressure.

A study of flow over a submerged barb by Fox et al., (2005) examined the time series of Reynolds stress as a method to investigate the presence and time scale of eddies of varying size. The results indicated the presence of large-scale macroturbulent eddies throughout the test section. While smaller-scale bed derived eddies were most pronounced in the deflected flow region where the barb had less influence on the flow, multiple small scale eddies including ejection and wake associated eddies persisted in the downstream overtopping and wake regions of the barb obstacle. These turbulent structures add temporal and spatial variability to flow where

the growth and decay of turbulent structures or eddies is intermittent and different eddy sizes can be identified by distinct temporal frequencies.

The influence of flow variability triggered by single obstructions is dependent on the level of submersion. Near bed small to medium scale flow structures often involve reverse, upward and downward velocities, negative pressure and 3D vortex shedding. Fully submerged obstructions in deep flow create strong local complex flows. Fully submerged obstructions that are moderately close to the surface interact with the free surface and can create standing waves. However, the influence of single flow obstructions seems to be localized as log velocity profiles can be measured within a distance upstream, downstream, and transverse to the obstruction. The arrangement of multiple flow obstructions would be expected to add another layer of complexity. Even under steady discharge conditions, the addition of flow obstructions increases the temporal variability of flow with the introduction of coherent turbulent and vortex structures. Additional flow variability is introduced with more complex structures such as large woody debris and with unsteady flows.

1.3.1. Hydraulic Modeling Limitations

Most hydraulic models designed for use with streams and rivers have similar data requirements. At a minimum, bed topography, roughness, and inflow/outflow boundary conditions serve as input to model velocity characteristics and water surface elevations. Typically these models are calibrated/validated using measured water surface, depth and velocity measurements. The level of detail in the calibration/validation stage of the modeling process differs depending on the number of modeled velocity dimensions. Simple 1D models used to represent hydraulic characteristics (depth and velocity) represent stream topography as a set number of cross sections with elongated cells consisting of variables averaged over the entire area covered by the cell. Data collection is limited to depth, flow, and substrate at cross sections. The selection of representative cross sections is important to adequately represent the river. More cross sections equate to more modeled detail.

To capture the transverse flows and spatial variability in of flows found in natural streams, a higher order (2D or 3D) hydraulic model is necessary. When moving from 1D to 2D models, data requirements increase significantly. Bed topography must be represented in greater detail to capture complex flow patterns. In addition, hydraulic characteristics (water surface,

depth, and velocity measurements) used to calibrate the model must also be spatially distributed. Two-dimensional hydraulic models apply basic principles of conservation of mass and momentum to a prismatic vertical water column bounded by the bed and a free surface leading to an assumption of hydrostatic pressure distribution. The major assumption of 2D models is depth-averaged flow. This assumption limits the utility of 2D models to velocity in the horizontal direction. These models do not account for vertical velocity or acceleration. The acceptability of depth-averaged flows needs to be considered with respect to the relative importance of vertical motions.

In addition, many 2D models do not have the ability to satisfy the non-slip boundary condition for flow around exposed boulders (Shen and Diplas, 2008). River2D (Steffler and Blackburn 2002), for example, handles wetting and drying of individual elements by coupling flow calculations with a groundwater model that calculates a common free surface aboveground in the channel and belowground on dry land. Other models turn cells on and off and insert no-flow boundaries based on a minimum depth criterion or change fluid properties at very small depths so that a very thin layer of fluid is always present. With a real fluid flow non-slip conditions should be satisfied at any solid boundary rendering the velocity equal to zero at the corresponding locations (Shen and Diplas, 2007).

The use of a 3D hydraulic model would help to resolve some of the issues with 2D hydraulic modeling. The influence of submerged rocks on flow complexity could be examined and vertical flow characteristics especially in local areas with high slopes could be more adequately predicted. The major drawback (and the major advantage) of utilizing a 3D hydraulic model to model flow in natural streams is the increase in model complexity. An increase in model complexity leads to an increase in data requirements both as input and to calibrate and validate the model. In general, the bed topography measurements for 2- and 3D hydraulic models are identical for similar horizontal scales of interest. The difference between the two models is only applied to bed topography when undercut boulder geometry or large woody debris is introduced into the model. To compare 3D hydraulic model results to measured data, the velocity needs to be measured at multiple points in the velocity profile.

Theoretically, a 3D hydraulic model would better represent the physical flow conditions created by complex geometry. The assumption of depth-averaged flow profiles limits the suitability of 2D models to flows without a strong vertical component. The inability to explicitly

model a non-slip boundary condition is a major conceptual drawback to utilizing 2D models to examine flow complexity. In addition, the eddy viscosity assumption is inferior to other two-equation turbulence closure models (Shen and Diplas, 2008).

Much of the error in hydraulic model predictions is associated with errors in topographic representation (Pasternack et al. 2006). Emerging technologies such as aerial and terrestrial LIDAR (light detection and ranging) show promise for obtaining high resolution, high accuracy topographic models but there are issues relating to reflections (or lack of reflection) from water surfaces and vegetation (Marchamalo et al. 2007; Reusser and Bierman 2007; Aggett and Wilson 2009). The large amount of data collected from these technologies result in a trade off between spatial coverage and morphologic detail (Heritage and Hetherington 2007), but aerial LIDAR can provide the data necessary to model long fluvial segments in which the data collection from tradition survey would be prohibitive (Marchamalo et al. 2007). Despite error associated with the lack of return through the water surface aerial LIDAR surveys have been combined with 2D modeling to model flood routing (Hunter et al. 2008; Lim and Cheok 2009); however the use of LiDAR for river flow modeling requires a series of post-processing steps including merging the LiDAR points with suitable bed data (Mandlburger et al. 2009)

1.4. Summary

At the reach scale, common in-stream habitat restoration or improvement activities include the addition of boulders and large woody debris (LWD), and riparian management (Bernhardt et al. 2005). It is difficult to quantify the success of these habitat restoration projects since the success of a restoration project is dependant on the specific goals of the project. There is evidence, however, that goals of increased fish biodiversity and density are not always met by changes to the physical habitat in the stream (Bond and Lake, 2003; Lepori et al., 2005b; Thompson, 2006). While the hierarchical relationships between habitat variables and biological relevance at a range of spatial scales are poorly understood, local habitat variables including depth, velocity, substrate, and temperature are often related to fish populations at a local scale. While it is likely that there are other factors inhibiting fish populations, such as water quality (e.g. Bond and Lake, 2003), there is a need for a better understanding of the relationship between stream restoration practices and the resulting changes to in-stream habitat. The literature related to the flow characteristics of in-stream structures is limited (e.g. Biron et al., 2005), although the biological importance of flow characteristics is assumed. Recent studies (e.g. Lepori et al., 2005b; DeJalon and Gortazar, 2006) indicate the need for more research to define habitat characteristics important to hydraulic modeling incorporating temporal variability, habitat heterogeneity and spatial structure of habitat patches.

1.5. *Dissertation Organization*

This dissertation follows a journal article format. Following the literature review (Chapter 1), three articles for publication are presented, each with its own abstract, literature review, results, discussion, conclusion and reference sections. Chapter 2 has been submitted and published by the Journal of Hydraulic Engineering. Chapters 3 and 4 are in preparation for publication by the authors. The final chapter, Chapter 5, provides general conclusions and areas for future research. The dissertation citation format follows that of the American Society of Civil Engineers (ASCE) Journal of Hydraulic Engineering. While this dissertation is a collection of three separate articles, together, these papers provide quantitative means to examine stream restoration practices for habitat improvement, specifically in-stream habitat evaluation and riparian management.

Chapter 2. Hydraulic Complexity Metrics for Evaluating In-Stream Brook Trout Habitat

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Chapter 3. Spatial Classification of Flow Refugia for Evaluating Brook Trout Habitat

Kozarek, J. L., J. P. Resop, W. C. Hession, and C.A. Dolloff. (In Prep).

Chapter 4. Combined Influence of Bank Vegetation and Watershed Imperviousness on Stream Temperature

Kozarek, J. L., W.C. Hession, J. D. Newbold, T. E. Johnson, B. W. Sweeney, and N. G. Johnson. (In Prep).

1.6. References

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Chapter 2: Hydraulic Complexity Metrics for Evaluating In-Stream Brook Trout Habitat

Kozarek, J. L., W. C. Hession, C. A. Dolloff, and P. Diplas. (2010). "Hydraulic complexity metrics for evaluating in-stream brook trout habitat." *Journal of Hydraulic Engineering*, 136(12)1067-1076. This manuscript is used with permission from ASCE.

2.1. Abstract

A two-dimensional hydraulic model (River2D) was utilized to investigate the significance of flow complexity on habitat preferences of brook trout (*Salvelinus fontinalis)* in the high-gradient Staunton River in Shenandoah National Park, VA, USA. Two 100-m reaches were modeled where detailed brook trout surveys (10-30 m resolution) have been conducted annually since 1997. Spatial hydraulic complexity metrics including area-weighted *circulation* and *kinetic energy gradients* were calculated based on modeled velocity distributions. These metrics were compared to fish density in individual habitat complexes (10-30 m sub-reaches) to evaluate relationships between fish location and average flow complexity. In addition, the fish density was compared to additional habitat variables including percent cascade, pool and riffle, and in-stream and riparian cover. There were negative correlations between modeled mean velocity and maximum depth and fish density; however, there were no statistically significant correlations between *kinetic energy gradients* or area-weighted *circulation* and fish density. Fish density was negatively correlated to in-stream cover and positively correlated to the percent of the channel dominated by protruding boulders and percent cascade. The structural complexity of cascade habitat and areas with protruding boulders create complex flow patterns indicating that flow complexity plays an important role in brook trout habitat preferences at the local scale. Linear discriminant analysis was used to further investigate the relationships between habitat variables and fish density. Using backward stepwise variable selection, the final explanatory model contained the percent protruding boulder, in-stream cover, maximum depth, percent pool, and mean velocity variables. These observations indicate that at a coarse spatial scale hydraulic

complexity may be an important component in fish habitat preferences; however, other habitat variables cannot be ignored and the hydraulic complexity metrics calculated using 2D modeling results were not explanatory. While spatial hydraulic complexity metrics provide quantifiable measures for evaluating stream restoration project impacts on in-stream habitat quality, the relationships between fish density and hydraulic complexity were not straightforward. This is likely due, in part to modeling limitations in this high-gradient, complex stream. Further research is needed at a range of spatial scales, stream types, and fish species to fully investigate the use of hydraulic complexity metrics to quantify in-stream habitat.

Subject Headings: Aquatic habitats, Hydraulic models, Two-dimensional models, Mountain streams, Boulders

2.2. Introduction

Interest in river and stream restoration has increased dramatically over the last two decades. Conservative estimates place river restoration costs for the continental U.S. in excess of $14 billion since 1990 with more than $400 million spent on restoration projects in the Chesapeake Bay Watershed (CBW) alone (Bernhardt et al. 2005; Hassett et al. 2005). The most commonly stated restoration goals include water quality, riparian management, in-stream habitat, fish passage, and bank stabilization (Bernhardt et al. 2005). This research focuses on the interdisciplinary linkage between aquatic ecology and engineering that must occur for successful stream restoration by addressing the lack of consistent methods to evaluate stream restoration projects. Understanding the relationships between structural complexity and hydraulic complexity will result in quantifiable metrics for evaluating stream restoration projects' impacts on in-stream habitat quality. The overall goal of this research was to assess the hydraulic characteristics of in-stream habitat that need to be considered for successful stream habitat restoration. The specific goals of this research were to quantify in-stream hydraulic complexity using metrics to describe flow structure; quantify the relationship between hydraulic complexity and fish habitat preferences; and ultimately, using metrics determined to be biologically relevant, evaluate in-stream habitat structures for their ability to create preferred hydraulic conditions for fish.

In-stream habitat refers to the physical habitat, or "living" space, of in-stream biota that encompasses the channel's physical structure, and the spatial and temporal dynamics of the flow regime (Maddock 1999). The most important reach-scale abiotic factors that affect fish in running water are temperature (directly and indirectly through oxygen consumption), rate of flow and fluctuation in discharge, and availability of suitable shelter (substratum) (Hynes 1970); however, mesohabitat characteristics including depth, velocity, substrate, and cover are important to fish for spawning, feeding, and refugia during high flows (Maddock 1999; Rempel et al. 1999; Schwartz and Herricks 2005; Smith et al. 2006).

In many ecological systems, biotic diversity is positively correlated with habitat heterogeneity (Bell et al. 1991; Rosenzweig 1995). Habitat heterogeneity in streams has been shown to be related to macroinvertebrate taxon richness, fish species diversity and density,

periphyton, and processing of coarse particulate organic matter (Gorman and Karr 1978; Biggs and Stokseth 1996; Brown 2003; Lepori et al. 2005b). In-stream habitat heterogeneity can refer both to substrate and flow characteristics such as depth or velocity. The spatial heterogeneity (or variability) of flow is associated with the hydraulic complexity within a stream. Aquatic organisms often inhabit and utilize complex flow patterns such as eddies, transverse flows, and velocity gradients (Fausch and White 1981; Hayes and Jowett 1994; Biggs et al. 1997; Rempel et al. 1999), while other complex hydraulic characteristics such as turbulent kinetic energy have been found to predict salmonid density (Smith et al. 2006).

Tritico and Hotchkiss (2005) evaluated turbulence parameters downstream of natural boulders in gravel-bed streams with varying degrees of roughness. Their findings, using isolated boulders, confirm that natural boulders create vertically-oriented vortex structures. The kinetic energy in these flow structures can be utilized by swimming fish (Videler et al. 1999; Enders et al. 2003; Liao et al. 2003). In addition, locally-accelerated streamwise velocity around obstructions provides favorable migration corridors for juvenile salmonids as it provides mean velocities that are greater than average reach velocities with low relative turbulence (Tritico and Hotchkiss 2005).

To describe in-stream hydraulic conditions with potential biological importance, metrics were proposed by Crowder and Diplas (2000; 2002; 2006) including *vorticity*, *circulation*, and *kinetic energy gradients* in an effort to quantify the flow variability that many species exploit. *Vorticity* (ξ) is a point metric that represents twice the rate that a fluid element rotates about its vertical axis (for 2D flow):

$$\xi = \left(\frac{\partial v}{\partial x} - \frac{\partial u}{\partial y} \right) \qquad (1)$$

where v is velocity in the x direction and u is velocity in the y direction. Vorticity created by exposed boulders and spur-dykes can generate important habitat such as scour holes (Shields et al. 1995) and can be exploited by macroinvertebrates feeding on drifting material (e.g. Way et al. 1995). The modified *circulation* metric (ξ_{AVE}; CRC) proposed by Crowder and Diplas (2000)

is the average absolute vorticity per unit area and represents a means to quantify area-weighted flow complexity:

$$\left|\xi\right|_{AVE} = \frac{\iint_{A_{tot}} \left|\xi\right| dA}{A_{tot}} = CRC \qquad (2)$$

where ξ is the absolute value of the vorticity and A_{tot} is the area of the region of interest. *Kinetic energy gradients* (KEG) quantify the spatial rate at which a flow's kinetic energy is changing around a point (or the average spatial change in kinetic energy between two points) and may be used to describe salmonid feeding locations where fish rest in relatively slow-moving water that is adjacent to faster water that transports food (e.g. Crowder and Diplas 2006). The KEG between two points can be estimated as:

$$\left|\frac{\partial}{\partial x}\frac{V^2}{2}\right| \cong \left|\frac{2V_{AVE}\frac{V_2 - V_1}{\Delta x}}{V_1^2}\right| = KEG \qquad (3)$$

where V is the velocity magnitude, and Δx is the distance between two points. Shields and Rigby (2005) have used the flow complexity metrics developed by Crowder and Diplas (2000; 2002; 2006) to discriminate between flow patterns upstream and downstream of an obstruction. However, no studies have been conducted to evaluate whether or not these metrics are biologically relevant at different spatial scales. Specifically, no studies address the questions: 1) Do fish prefer flow structures identified by flow complexity metrics? 2) Can these metrics be used to predict the location of fish populations? and 3) Can they be used to guide or evaluate stream habitat restoration? This study addresses the last two questions at a coarse (>10 m) spatial scale.

Several hydraulic models have been used to simulate in-stream habitat suitability. The Physical Habitat Simulation Model (PHABSIM) is a well-established hydro-ecological model that provides a suite of tools for the numerical modeling of hydraulic habitat suitability for fish

and invertebrate species based on field measurements of channel slope, water depth, velocity, and substrate (Bovee 1982; Maddock 1999; Booker and Dunbar 2004); however, PHABSIM is one-dimensional (1D) and averages velocity, substrate, and depth values between measured cross sections and cannot account for the range of habitat types or physical conditions adjacent to a location within a stream (Maddock 1999; Waddle 2001). Two- and three-dimensional (2D and 3D) hydraulic models have been used to calculate flow characteristics in streams as a measure of habitat suitability (Booker 2003; Crowder and Diplas 2006; Clark et al. 2007; Shen and Diplas 2008). The advantage of 2D and 3D habitat models over the conventional 1D models (e.g. PHABSIM) is the ability to spatially determine depth, velocity, and flow direction, allowing the user to evaluate areas of particular ecological importance, such as refugia from high flows (Shen and Diplas 2008). Similar to PHABSIM, a 2D hydraulic model such as River2D (Steffler and Blackburn 2002), can be combined with habitat suitability indices (HSI) for target species to predict the weighted usable area (WUA), or relative amount of preferred habitat available to a target species (Ghanem et al. 1996; Steffler and Blackburn 2002). However, WUA ignores local flow complexity and structure and predicts habitat suitability based on single-point flow and depth requirements for a particular species while in reality, the spatial distribution of WUA provides a more realistic view of habitat within the stream. Previous research using River2D found reach-averaged WUA did not correlate to fish populations and the distribution of usable habitats was a better determining factor (Clark 2006).

Channel alterations, such as straightening and removal of flow obstructions, result in a loss of physical habitat diversity. Traditionally, engineers concerned primarily with flood mitigation, eliminated complexity from streams. In reverse, many restoration projects aim to restore some level of complexity to streams. Common in-stream habitat restoration or improvement activities include the addition of boulders and large woody debris (LWD; Bernhardt et al. 2005). Other stream restoration practices such as weirs (low dams) and deflectors are used for habitat enhancement (Biron et al. 2005; DeJalon and Gortazar 2006). It is difficult to quantify the success of these habitat restoration projects since the success of a restoration project is dependent on the specific goals of the project. There is evidence, however, that goals of increased fish biodiversity and density are not always met by changes to the in-stream physical habitat (Bond and Lake 2003; Lepori et al. 2005a; Thompson 2006). While it is probable that there are other factors inhibiting fish populations, such as water quality (e.g. Bond

and Lake 2003), there is a need to better understand the relationship between stream restoration practices and the resulting changes to in-stream habitat. The literature related to the flow characteristics created by in-stream structures is limited (e.g. Biron et al. 2005), although the biological importance of flow characteristics is assumed. Recent studies (Lepori et al. 2005b; DeJalon and Gortazar 2006) indicate the need for more research to define habitat characteristics important to hydraulic modeling and stream restoration.

2.3. Methods

2.3.1. Study Site

The Staunton River is a second-order headwater stream originating on the eastern slope of the Blue Ridge Mountains in Shenandoah National Park (SNP), VA (Figure 2-1). This river flows approximately 6.5 km to its confluence with the Rapidan River and drains approximately 11 km^2 (Hyer et al. 1995). The average channel width is 3.5 m and the average channel gradient is 10%. The channel consists of pools separated by step-pool cascades, small (<2 m) waterfalls, and bedrock slides and has been subject to long-term flow and water-quality monitoring (Department of Environmental Sciences at the University of Virginia, Charlottesville). There are two primary fish species in the Staunton River: brook trout *Salvelinus fontinalis* and blacknose dace *Rhinichthys atratulus*. In June 1995, a debris flow completely eliminated brook trout from the affected 1.9 km of the Staunton River and removed trees from a 30-m band in the riparian area (Roghair et al. 2002). The June 1995 storm and resulting debris flow are described in Karish et al. (1997).
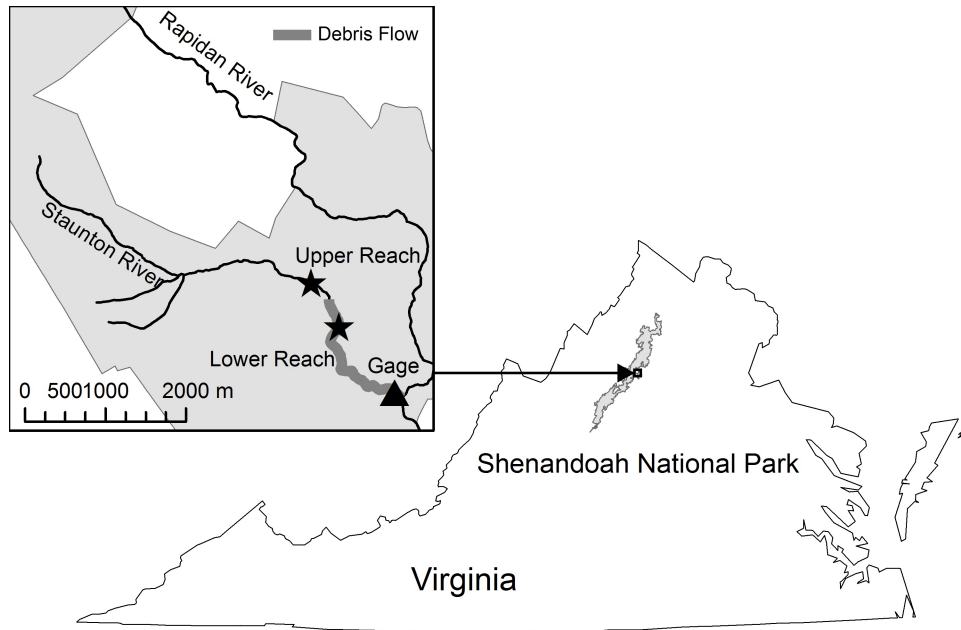
Figure 2-1. Staunton River is located in Shenandoah National Park

2.3.2. Brook Trout Population Density

This study focused on brook trout (*Salvelinus fontinalis),* a valuable native game fish in Virginia. We evaluated hydraulic complexity in two 100-m reaches on the Staunton River. The Staunton River was chosen as a study site because there are long-term (11 years) and high-resolution (sampled every 10-30 m) brook trout data available from previous studies. The brook trout data used in this study was collected as part of a recolonization and post-recolonization brook trout movement study (Roghair et al. 2002; Roghair and Dolloff 2005). Beginning in 1997, brook trout were sampled biannually (May and October) in a continuous 1 km reach of the Staunton River: 575 m in the debris flow affected area and 390 m in the unaffected area of the stream (above the debris flow). The total brook trout abundance data set used in this study ranges from 1997-2004 in autumn (mid-October) and from 1997 to 2007 in spring (mid-May to mid-June). To locate fish, the river was subdivided into habitat complexes (10-30 m sub-reaches comprised of multiple pools and riffles and terminating at potential low flow barriers). Brook trout were captured by making a single pass through each habitat complex with a backpack electrofishing

unit (73% fist pass capture efficiency; Roghair 2000). The length (mm) weight (g) and location of capture (habitat complex) were recorded for each fish (Roghair et al. 2005). From this larger data set, brook trout abundance data for two modeled 100-m reaches were used in our study to evaluate hydraulic characteristics of brook trout habitat preferences. The lower reach was within the debris flow affected area, and the upper reach was approximately 600-m upstream. The study reaches were chosen using the following criteria: at least five habitat complexes, no tributaries over the study reach, single channel, no major waterfalls or wood jams. The average channel gradient within the study reaches is 7%.

2.3.3. Habitat Survey

To characterize habitat within each reach, a field survey was conducted to measure in-stream cover (ISC) riparian cover (RC), large woody debris (LWD) and areal percent of pool (PL), riffle (RF) or cascade (CS) in each habitat complex. Each habitat complex was visually divided into percent pool, riffle, and cascade habitat. Riffles and pools were identified using descriptions similar to those in Gordon et al. (2004). Cascades were steeper areas with larger (boulder) bed material. The areal percent of the stream channel dominated by protruding boulders (BD) was visually estimated. Riparian cover (RC) was measured within each habitat complex using a convex densitometer. Four densitometer measurements, one each facing upstream, left bank, right bank, and downstream were averaged. The amount of in-stream cover (ISC) within each habitat complex was estimated in the field by measuring the horizontal width and length of undercut boulder, banks, or overhanging vegetation. The volume of large wood (> 1 m in length and > 10 cm diameter) within the channel was quantified and divided by the area of each habitat complex to get an area weighted volume. The measured in-stream cover area and large wood volume were normalized by the length of each habitat complex (ISC_N; LWD_N). Reach-wide pebble counts were conducted to characterize bed substrate (Wolman 1954). Substrate samples were collected in each reach on two different days with 100 pebbles collected in a representative riffle and 100 pebbles collected in a representative pool. On the second sampling date, a substrate sample (100 random pebbles) was collected in a transition area (or a segment of stream that was not classified as pool or riffle).

2.3.4. Hydraulic Modeling

A 2D hydraulic model (River2D) was utilized to calculate the spatial distribution of velocity vectors needed to calculate flow complexity metrics such as those described by Crowder and Diplas (2006) following the procedure outlined by Steffler and Blackburn (2002). The metrics were compared to spatially and temporally extensive fish data to determine which are relevant to habitat preferences.

Typical input to River2D includes bed topography, initial roughness estimates, discharge at the upper cross-section, and water surface elevations at the downstream cross-section (Steffler and Blackburn 2002). Bed topography was measured in each 100-m reach by detailed electronic total station surveys (> 2400 points per reach; > one point per 0.5 m^2). To accurately represent complex topography, the shape of each boulder was captured by surveying the apex(es) of large boulders and surveying a minimum of four points around the base. Initial bed roughness values were obtained from standard pebble counts. A finite element triangular computational mesh consisting of approximately 100,000 nodes was created for each reach.

The primary calibration factor for River2D is the roughness factor and the model was calibrated by minimizing the mean absolute error (MAE) between measured depth and velocity values and model output (Lacey et al. 2004). Velocity was measured using a 3D acoustic Doppler velocimeter (SonTek FlowTracker Handheld ADV) at three cross-sections and a minimum of 25 additional points throughout each reach (right, left, and center every 5-10 m). Velocity measurements with high numbers of spikes, high signal to noise ratio (SNR) variation, or with boundary condition interference were either repeated in the field or removed prior to calibration. Each velocity, depth, or water surface measurement was surveyed to overlay the measurement with the model results. The Shenandoah Watershed Study (SWAS) maintains a continuous discharge gage (1993-2006) on the Staunton River approximately 2 km downstream of the lower study reach near the confluence with the Rapidan (Figure 2-1). Flows in the study reach were calculated by adjusting the measured discharge by the watershed size. The discharge measurements corresponding to the brook trout sampling dates ranged from baseflow (0.02 cms) to 0.3 cms. The model was calibrated at the middle model flow (0.2 cms) within the lower reach. The calibration was then checked at high flow (0.8 cms) in the lower reach and low flow (0.1 cms) in the upper reach.

River2D allows adjustment flow options including the upwinding coefficient of the Petrov-Galerkin finite element scheme used to solve the hydrodynamic equations. Because this model was run as steady state, the default value of 0.5 was used. For this study, the groundwater flow options storativity and minimum depth for groundwater flow were also the default values. The transmissivity was changed to 0.05 from the default value of 0.1 to reduce the loss of flow to groundwater (Steffler and Blackburn 2002). This value was chosen to minimize the loss of flow to groundwater while not dramatically increasing the model run time. River2D models transverse turbulent shear stresses with a Boussinseq type eddy viscosity with three user definable coefficients (Steffler and Blackburn 2002). The first term, ε_1, is a constant that can be used to stabilize the solution for very shallow flows. The second term, ε_2, is an eddy viscosity bed shear parameter that typically ranges from 0.2 to 1.0. The default value suggested in the River2D manual is 0.5 (Steffler and Blackburn 2002). The third term, ε_3, represents the horizontal shear and can become important in deeper lake flows or flows with high transverse velocity gradients. In general, River2D is insensitive to values of ε_2 (Lacey et al. 2004). The sensitivity of this River2D model to these terms was evaluated by varying the values of ε and comparing to the calibration flow of 0.2 cms in the lower reach.

Typical output of River2D includes depth, water surface elevation, and velocity in two dimensions (across and downstream) (Figure 2-2). The model was run at three different flows to coincide with select brook trout sampling dates, 0.1 cms, 0.2 cms, and 0.3 cms. Baseflow sampling dates were excluded from this analysis because of the instability of the model at low flows. Velocity vectors from each run were exported on a 0.10 m grid and used to calculate flow complexity metrics. The mean *vorticity* and *kinetic energy gradient* within each habitat complex were calculated, and the *vorticity* values were used to calculate the modified *circulation* (area-weighted metric that allows comparison of flow complexity between habitat complexes and reaches).
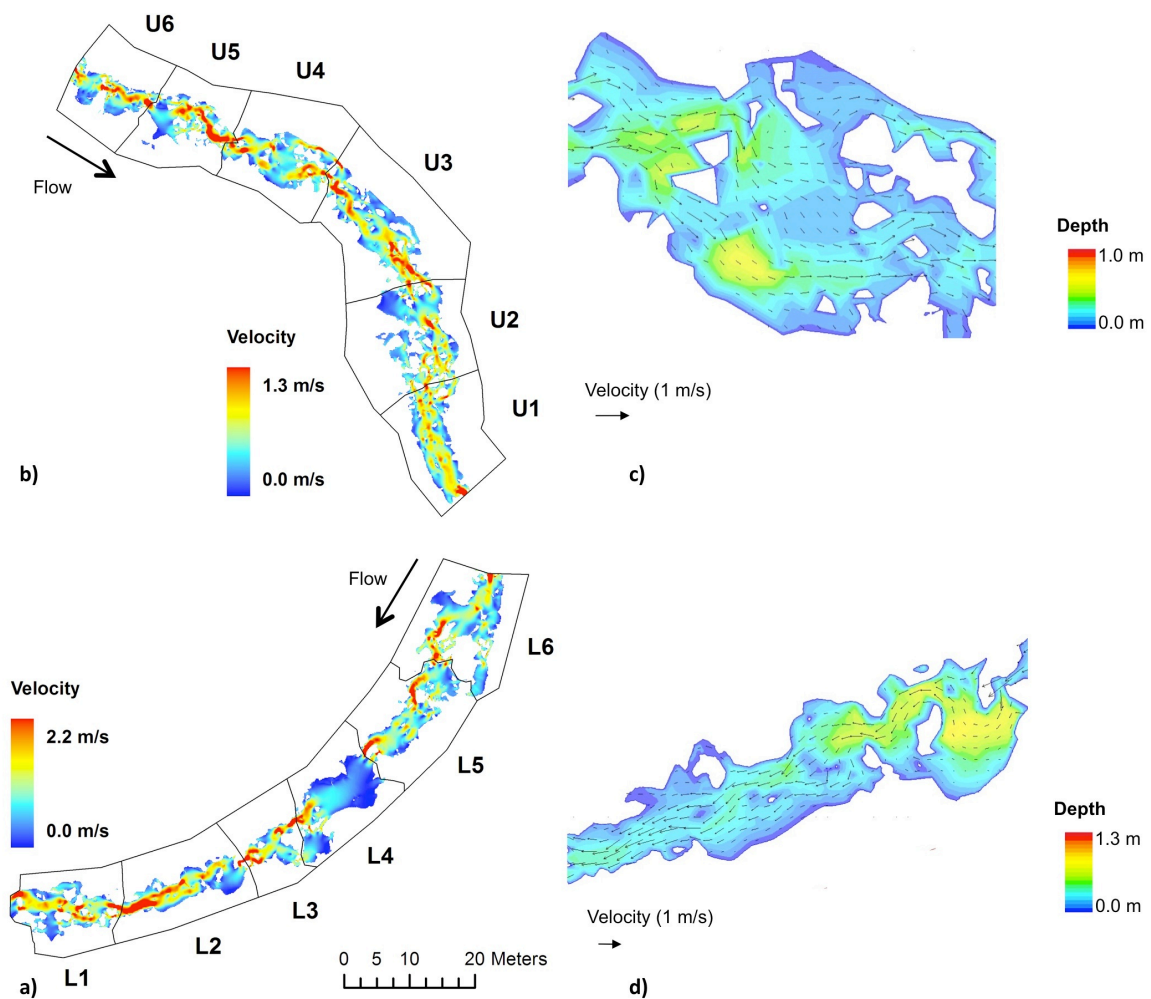
Figure 2-2. Modeled velocity in (a) lower; (b) upper reach for 0.3-m^3/s model run. Modeled velocity vectors and depth for the 0.3-m^3/s run for habitat complex: (c) U4; (d) L2 (see Table 2-1).

2.3.5. Statistical Analysis

Statistical analyses were performed on habitat variables and flow complexity variables between and among reaches. The habitat variables included RC (%), ISC$_N$ (m^2/m), LWD$_N$ (m^3/m), D$_{50}$ (mm), RF (%), PL (%), and CS (%). The analyses also included the areal percent protruding boulders estimated during the habitat survey BD (%). The flow variables used in

statistical analyses included area-weighted circulation (CRC), maximum depth (MAXD), mean velocity (VEL), and mean kinetic energy gradient (KEG).

To compare reach-level differences in fish, habitat, and flow variables, the groups were compared using SigmaPlot (Systat Software Inc. 2008). Variables that passed the normality and equal variance test were analyzed using a t-test. Variables that failed the normality or equal variance test were compared using a non-parametric Mann-Whitney Rank Sum test (Conover 1998). The results for the 0.2 cms modeled discharge were used to compare the differences in hydraulic complexity between reaches. Differences in brook trout abundance sampled in each habitat complex were analyzed using the non-parametric Kruskal-Wallis one-way analysis of variance on ranks (Ott and Longnecker 2001). Similarly, differences in brook trout abundance by sampling date were evaluated.

To examine the importance of the habitat and flow complexity variables on brook trout density, the habitat and hydraulic variables were compared to the brook trout density (# fish/area) for each reach individually and for the reaches combined. Spearman's Rank non-parametric correlation coefficients were calculated using JMP software (JMP; SAS Institute, Inc. 2007).

Finally, the brook trout data were divided into four groups based on quartiles ranging from low density (A) to high density (D). Stepwise discriminant analysis, a multivariate technique, was used to evaluate the ability to classify the brook trout into density groups based on hydraulic complexity and habitat variables. General trends in the variable means for the predicted classification groups were evaluated. This analysis was conducted using JMP software.

2.4. Results

2.4.1. Hydraulic Model Results

The calibration MAE at 0.2 cms for water surface elevations was 0.05% (0.05 m), 25% (0.06 m) for depth, and 48% (0.12 m/s) for velocity. At the 0.8 cms flow, the MAE for water surface elevations was 0.06% (0.06 m), 27% for depth (0.07 m), and 45% (0.14 m/s) for velocity values. These values are higher than other River2D calibration values reported in the literature (Lacey et al. 2004 and Hayes et al. 2007); however, these were the minimum absolute errors achieved by adjusting the roughness factor, k_s, and the eddy viscosity parameters and are

indicative of the issues of modeling complex systems such as the Staunton River. The model results were also checked at low (0.1 cms) flows on the upper reach resulting in a MAE of 0.07% (0.07 m), 30% (0.06 m) and 56% (0.12 m/s) for water surface elevation, depth, and velocity, respectively.

The model was most sensitive to the ε_2 eddy viscosity coefficient; changing this value to 0.2 and 1.0 resulted in a change in velocity values of 0.02% and 0.1%, respectively. In general, increasing this value resulted in faster flows behind obstructions, and slower flows mid-stream compared to the default value, while decreasing ε_2 resulted in slower flows behind obstructions and faster flows in mid-stream. Using the 0.2 cms flow in the lower reach to calibrate, the final eddy viscosity coefficients values were 1.0, 0.01, and 0.1 for ε_2, ε_1, and ε_3, respectively. The predicted depth and velocity for the 0.3 cms flow is shown in detail in Figure 2-2.

2.4.2. Reach Scale Comparison

Over the time period of this study, within the modeled reaches, the abundance of brook trout normalized by the length was significantly different between habitat complexes and sampling dates (Mann-Whitney Rank Sum test; $p < 0.001$; Figure 2-3 and Figure 2-4); however, there was no significant difference in brook trout abundance between seasons (spring and fall) or between reaches (lower and upper) (Figure 2-5).

Differences between the two reaches were most evident in the habitat survey results. The median value for the upper reach was 87.8% RC, while the median value for the lower reach was 54.9% (Table 2-1). These values are significantly different ($p = 0.002$; Mann-Whitney Rank Sum Test). The mean ISC_N was 0.38 m/m in the lower reach and 0.17 m/m in the upper reach; these values were also significantly different ($p = 0.011$; t-test). The percent of each habitat complex characterized as pool was also significantly different between reaches ($p = 0.027$; t-test). The mean in the lower reach was 50.8% covered by pools and 28.3% of the wetted area in the upper reach was characterized as pool. In general, the substrate in the debris flow lower reach was larger than in the upper reach (Table 2-2). The LWD_N, RF, CS and BD habitat variables were not significantly different between reaches. In addition, the hydraulic variables, MAXD, VEL, KEG, and CRC were not significantly different between reaches (Table 2-3).
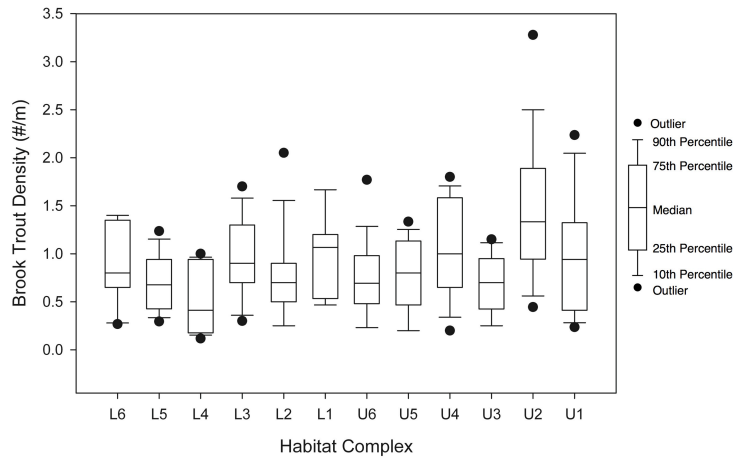
Figure 2-3. Brook trout caught per unit length of stream in each habitat complex combining all sampling dates (Fall 1997 - 2004 and Spring 1997 - 2007)
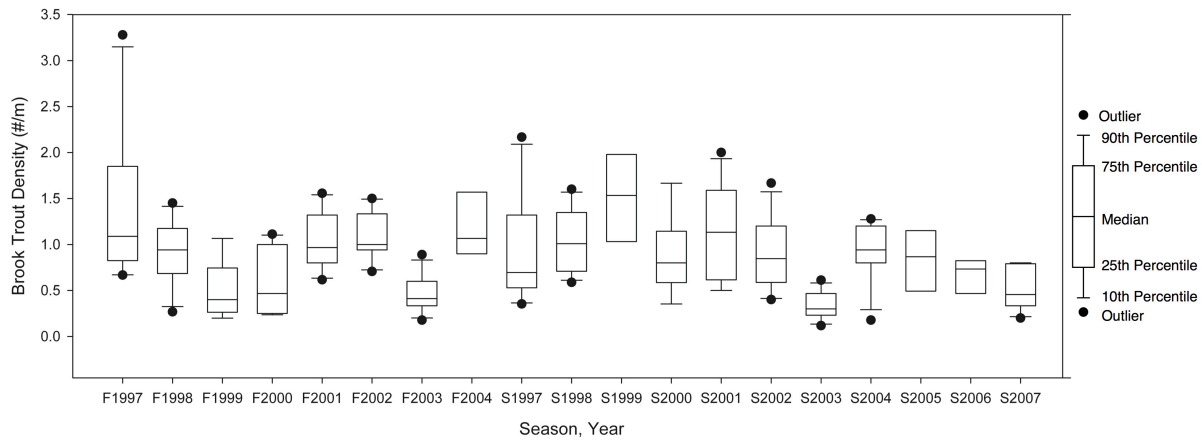
Figure 2-4. Brook trout per unit stream length for each sampling date for all habitat complexes within the study reaches. F signifies sampling dates in fall (October) and S signifies spring sampling dates (late May/early June)
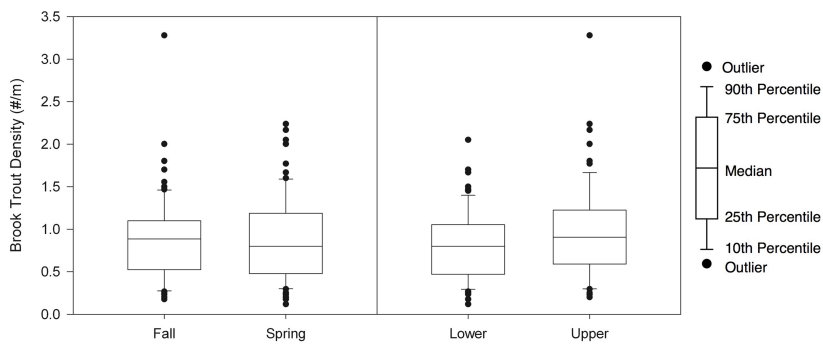
Figure 2-5. Brook trout caught per meter stream length: (a) during fall and spring sampling; (b) in upper and lower reaches on the Staunton River

Table 2-1. Habitat survey variables and mean brook trout counts for each habitat complex on the lower and upper study reaches of the Staunton River.

Habitat complex[a]	Length (m)	Riparian Cover (RC)	In-stream Cover (ISC_N) (m^2/m)	LWD_N[b] Volume (m^3/m)	Riffle (RF)	Pool (PL)	Cascade (CS)	Boulders (BD)	# Brook Trout/m (mean ± SD)
L1	15	38.5	0.45	0.01	40	40	20	50	0.96±0.41
L2	20	80.3	0.31	0	20	60	20	50	0.75±0.49
L3	10	41.3	0.20	0	10	50	40	50	0.89±0.44
L4	17	49.9	0.44	0	10	80	10	30	0.52±0.34
L5	17	59.8	0.55	0.01	30	50	20	30	0.66±0.33
L6	15	60.8	0.33	0.02	60	25	15	50	0.85±0.38
U1	17	86.8	0.10	0.02	70	20	10	10	1.01±0.59
U2	18	84.8	0.12	0.06	40	40	20	44	1.43±0.69
U3	20	85.3	0.29	0	60	30	10	40	0.68±0.29
U4	15	92	0.33	0.02	35	15	50	55	1.05±0.49
U5	15	88.7	0.09	0.10	40	25	35	65	0.78±0.36
U6	13	90.5	0.11	0	45	40	15	30	0.75±0.39

[a] Habitat complexes were numbered progressing upstream in each reach. U refers to the upper (unaffected) reach. L refers to the lower (affected) reach.

[b] LWD_N is the volume of large woody debris measured per unit stream length.

Table 2-2. Substrate distribution in the lower and upper reaches of the Staunton River.

	Upper			Lower			Combined
	Pool	Riffle	Total	Pool	Riffle	Total	Total
D_{50}[a] (mm)	11	22	13	35	44	33	19
D_{84}[b] (mm)	150	160	150	300	310	300	270

[a] Median particle size

[b] 84th percentile particle size

Table 2-3. Modeled hydraulic variables (0.2 cms) for each habitat complex on the lower and upper study reaches of the Staunton River.

Habitat Complex	Flow 0.1 m³/s				Flow 0.2 m³/s				Flow 0.3 m³/s			
	Maximum Depth (MAXD) (m)	Mean Velocity (VEL) (m/s)	Mean KEG (m⁻¹)	Area-Weighted Circulation (CRC) (s⁻¹)	Maximum Depth (MAXD) (m)	Mean Velocity (VEL) (m/s)	Mean KEG (m⁻¹)	Area-Weighted Circulation (CRC) (s⁻¹)	Maximum Depth (MAXD) (m)	Mean Velocity (VEL) (m/s)	Mean KEG (m⁻¹)	Area-Weighted Circulation (CRC) (s⁻¹)
L1	0.50	0.16	7.9	0.40	0.57	0.23	8.1	0.53	0.61	0.29	7.5	0.61
L2	0.69	0.17	9.0	0.40	0.78	0.24	5.9	0.49	0.85	0.30	6.4	0.50
L3	0.29	0.17	6.3	0.43	0.55	0.25	6.7	0.61	0.62	0.33	5.8	0.81
L4	1.12	0.07	8.3	0.12	1.22	0.11	7.8	0.18	1.27	0.14	6.9	0.26
L5	0.42	0.14	7.0	0.30	0.50	0.21	7.0	0.44	0.56	0.27	8.2	0.64
L6	0.41	0.12	10.9	0.26	0.50	0.19	6.3	0.36	0.55	0.24	6.1	0.46
U1	0.28	0.18	6.6	0.37	0.34	0.25	6.6	0.51	0.39	0.29	7.0	0.64
U2	0.43	0.12	7.3	0.31	0.51	0.19	7.7	0.45	0.55	0.23	6.6	0.51
U3	0.41	0.15	7.4	0.42	0.48	0.23	9.2	0.47	0.52	0.29	6.4	0.55
U4	0.44	0.16	6.3	0.35	0.55	0.22	5.0	0.45	0.59	0.28	6.4	0.53
U5	0.84	0.14	7.9	0.45	0.94	0.24	7.5	0.66	0.98	0.30	7.8	0.69
U6	0.45	0.16	10.5	0.51	0.56	0.23	7.8	0.65	0.62	0.28	9.6	0.62

The debris flow affected reach (lower) had more pool area, more in-stream cover, and less riparian cover. The debris flow event (June 1995) removed all vegetation from a 30-m band in the riparian area (Roghair et al. 2002); therefore, the vegetation in the lower reach was a maximum of 13 years old at the time of the riparian cover measurements. It is likely that the difference in in-stream cover between reaches was also due to the effects of the 1995 debris flow. The majority of the measured in-stream cover was attributed to the underside of boulders within the channel. In the upper reach, these spaces have been filled in with sediment, while the lower reach is most likely still redistributing boulders from the debris flow. The hydraulic variables were not significantly different between reaches; however, these variables were calculated with a 2D model that cannot account for 3D flow structure that could occur due to complex topography such as the underside of boulders.

2.4.3. Local Scale Comparison

Since we found no significant difference between the abundance of brook trout sampled in each reach, the results were combined to examine the habitat variables' influence on brook trout habitat preference (the number of brook trout sampled in each habitat complex) using a non-parametric correlation analysis. To evaluate habitat complexity, the brook trout density was compared to MAXD, VEL, KEG, and CRC within each habitat complex (Table 2-4). The brook trout density was not significantly correlated ($\alpha = 0.05$) to the KEG or CRC when both reaches were combined, but was negatively correlated to both the MAXD and VEL metrics. For the habitat variables, the strongest significant ($\alpha = 0.05$) correlated variable was CS (Spearman's $\rho = 0.35$) followed by BD ($\rho = 0.23$) and ISC_N ($\rho = -0.19$).

Table 2-4. Spearman's rank correlation coefficients (ρ) and p-values comparing brook trout density to habitat variables. p-values less than 0.05 indicate significant correlations.

Variable	Combined		Lower		Upper	
	ρ	p-value	ρ	p-value	ρ	p-value
CS	0.3486	<0.0001	0.549	<0.0001	0.2287	0.0533
PL	-0.0888	0.3005	-0.3374	0.0056	0.1239	0.2998
BD	0.2292	0.0069	0.4229	0.0004	0.1233	0.3022
ISC_N	-0.1912	0.0247	-0.3368	0.0057	-0.0106	0.9298
RC	0.0351	0.6826	-0.0801	0.5225	0.0145	0.9039
RF	-0.0333	0.6979	0.016	0.8982	-0.2382	0.0439
LWD_N	0.1445	0.0908	0.0378	0.7634	0.2041	0.0855
MAXD	-0.4148	<0.0001	-0.6775	<0.0001	-0.1193	0.3183
KEG	-0.0529	0.5376	-0.0872	0.4862	0.0448	0.7087
CRC	-0.1071	0.2112	0.2143	0.084	-0.4257	0.0002
VEL	-0.2108	0.0131	0.1148	0.3586	-0.528	<0.0001

The area-weighted *circulation* (CRC) represents a means of calculating the 2D horizontal flow complexity. *Kinetic energy gradients* (KEG) quantify the average spatial change in kinetic energy between two points. In general, brook trout density tended to increase with increasing flow complexity. This was evidenced by positive correlations with the CS and BD metrics. The reverse was true in the upper reach where brook trout density was negatively correlated to the CRC metric (Figure 2-6). Further inspection of Figure 2-6 indicates that there may be a threshold in brook trout habitat complexity somewhere between approximately 0.3 and 0.5 s^{-1}. In this study, KEG metrics were calculated for each point from the surrounding points and the mean KEG for each habitat complex was not significantly correlated to fish density. Stronger spatial changes in kinetic energy would be found in complex flows such as flows around an obstruction (boulder). Future work should identify and quantify the relative habitat area of specific circulation zones (wakes behind boulders) within each habitat complex and evaluate KEG based on predictions from Crowder and Diplas (2006) for ideal feeding habitat for brook trout (KEG from 4 to 14 m^{-1}). In this study, the average KEG in each habitat complex fell within this range indicating that there was significant flow complexity as represented by KEG in every habitat complex. Fish density was negatively correlated to both average velocity (VEL) and maximum depth (MAXD) in each habitat complex. These relationships indicate that velocity or depth

characteristics are inadequate to represent brook trout habitat preferences in the complex flows created by boulders and other flow obstructions in Staunton River.
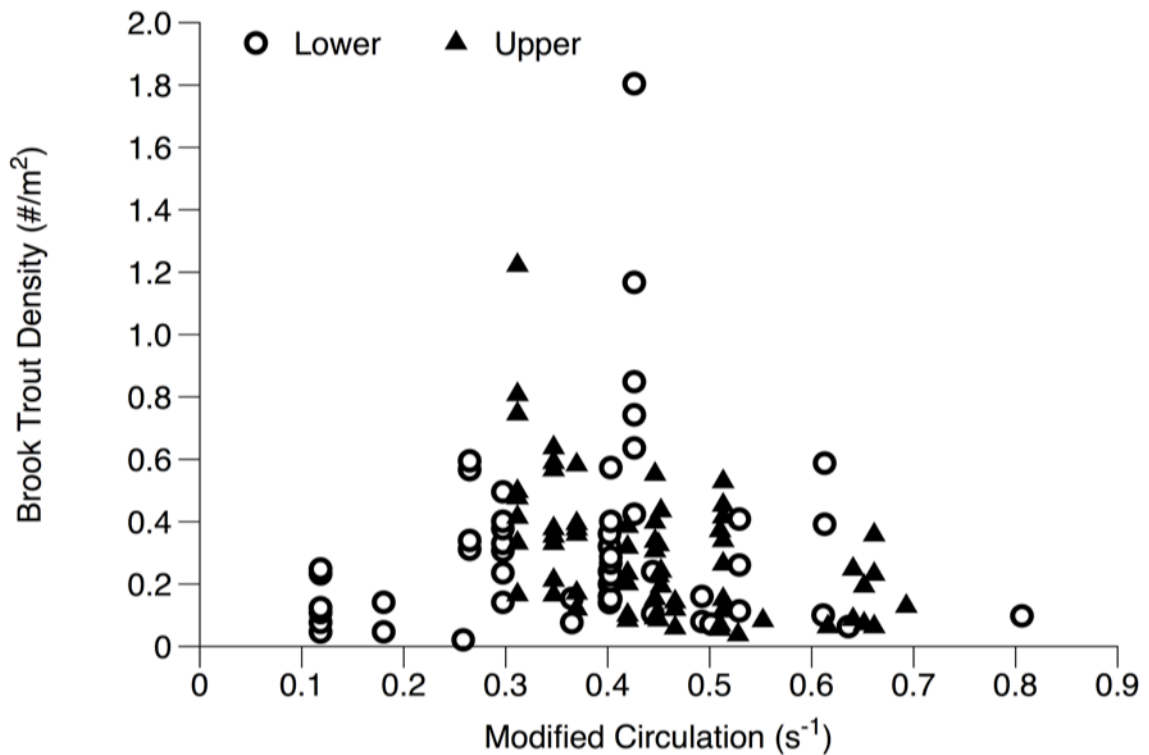
Figure 2-6. Brook trout density and area-weighted circulation (CRC) in the lower (debris-flow affected) and upper reaches of the Staunton River

2.4.4. Discriminant Analysis

The importance of the habitat variables was further analyzed using linear discriminant analysis to classify the brook trout density into categories. Because the % boulders (BD) was highly correlated to the CS, the BD variable was included in all discriminant analyses. The discriminant analysis using habitat variables (LWD_N, PL, ISC_N, RC, and BD) misclassified 58% of the fish density samples. Or in reverse, predicted classifications of brook trout density-based habitat variables were only 42% correct indicating that these variables alone were not able to predict brook trout habitat preferences. When the brook trout densities were misclassified based on habitat variables, the mean misclassification was 1.60 categories. General trends between group means for the predicted trout density categories indicated that brook trout density

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increased as in-stream cover decreased, PL decreased and the large wood volume (LWD_N) increased (Table 2-5).

Table 2-5. Predicted habitat variable means for fish density categories resulting from discriminant analysis.

Fish density category	% riparian cover (RC)	% pool (PL)	% boulder (BD)	In-stream cover (ISC_N)	Large wood (LWD_N)
A (low)	50	80	30	0.4	0
B	71	55	41	0.1	0
C	73	26	40	0.3	0.01
D (high)	77	39	47	0.1	0.04

Linear discriminant analysis was again performed using the modeled hydraulic variables. When compared to the discriminant analysis for only the habitat variables, there is more separation between the brook trout density categories when the discriminant analysis is conducted using the modeled hydraulic variables; however, the hydraulic variables alone did not adequately categorize the brook trout habitat preferences (misclassified 51%; mean misclassification 1.48 categories). In general, for the habitat complexes in this study, the brook trout density increased as the maximum depth and average velocity decreased (Table 2-6) as predicted by the correlation analysis. As there were no correlations between KEG, CRC, and brook trout density, the relationship between brook trout density and the trend in predicted means of these variables is also not straightforward. The mean *kinetic energy gradient* (KEG) had a maximum value at medium brook trout densities, while the area-weighted *circulation* metric had a minimum value at the same density. Although there is not much variation in the predicted means from the discriminant analysis, it is likely that there is a threshold flow complexity preferred by brook trout indicated by the lack of a straight forward correlation.

Table 2-6. Predicted hydraulic variable means for fish density categories resulting from discriminant analysis.

Fish Density Category	Maximum depth (MAXD; m))	Mean velocity (VEL; m/s)	Mean KEG (m^{-1})	Circulation (CRC; s^{-1})
A (low)	0.73	0.21	7.3	0.45
B	0.57	0.16	7.7	0.38
C	0.42	0.16	8.1	0.37
D (high)	0.48	0.15	7.8	0.41

The best discriminant analysis model combined both the habitat variables and the hydraulic variables. While the brook trout density was misclassified 45% of the time, the majority of the misclassifications were by one fish density category (mean misclassification of 1.35 categories). Using backward stepwise variable selection ($\alpha = 0.1$), the final model contained the BD, ISC$_N$, PL, MAXD, and VEL variables. While hydraulic complexity metrics are not represented in this model, the non-hydraulic variable (% boulder, BD) represents the structural complexity within the channel and certainly creates flow complexity; however, this flow complexity may be better represented using a 3D hydraulic model.

2.5. Conclusions

While our study was limited to one fish species in one stream, results indicate that metrics based on hydraulic engineering principles may be used to evaluate in-stream habitat at coarse spatial scales (10-30 m) but not without evaluating other non-hydraulic habitat parameters. The debris flow that disturbed the lower reach of this study site in 1995 appears to have lasting impacts on the stream structure and habitat. Although there were no significant differences in brook trout density between the reaches, there were differences in habitat variables, as well as which habitat variables were correlated to brook trout density. There was not a strong correlation among reaches between the CRC metric and brook trout density; however, there may be a threshold value for brook trout hydraulic complexity preferences (see Figure 2-6), but more data over a larger area and range of flows would be needed to confirm this hypothesis. While velocity characteristics and flow structure play a role in the brook trout habitat

preferences at the local scale (10-30 m), the relationships between variables are complex and difficult to predict as with any natural system.

Our results illustrate that hydraulic complexity metrics calculated from 2D hydraulic models can be useful in describing flow structure of in-stream habitat that cannot be modeled by 1D hydraulic models. However, the model did not accurately predict depth or velocity values particularly well because of the complex topography and steep slope of the stream. Therefore, the hydraulic modeling results of this study should not be used as predictive values, but as a tool to evaluate trends in hydraulic complexity and brook trout habitat preferences. This study spanned 11 years and included both spring and autumn sampling providing a unique opportunity to evaluate hydraulic complexity metrics integrated over time at a coarse scale; however, to evaluate the applicability of hydraulic complexity metrics for habitat modeling and stream restoration, individual habitat areas such as wakes behind boulders need to be recognized.

2.6. Acknowledgements

2.7. Notation

ξ	=	vorticity;
u, v	=	velocity in the x – and y –directions;
k	=	unit vector in the vertical direction;
$\lvert\xi\rvert_{AVE}$, CRC	=	area-weighted modified circulation metric;
A_{tot}	=	area of region of interest (i.e. habitat complexes);
KEG	=	average spatial change in kinetic energy between two points;
V	=	velocity magnitude;
Δx	=	distance between two points;

k_s = roughness height;

ε_1 = River2D eddy viscosity constant;

ε_2 = River2D eddy viscosity bed shear parameter;

ε_3 = River2D eddy viscosity horizontal shear parameter;

$MAXD$ = maximum depth in area of interest;

VEL = average velocity in area of interest;

LWD_N = large wood volume normalized by habitat complex length;

ISC_N = in-stream cover area normalized by habitat complex length;

PL = percent characterized as pool;

RF = percent characterized as riffle;

CS = percent characterized as cascade;

BD = percent of wetted channel dominated by protruding boulders

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Chapter 3: Spatial Classification of Flow Refugia for Evaluating Brook Trout Habitat

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3.1. Abstract

A spatial clustering scheme (Anselin local Moran's I) was utilized in conjunction with a two-dimensional hydraulic model (River2D) to investigate the significance of spatial distribution of hydraulic habitat variables on brook trout (*Salvelinus foninalis*) habitat preferences. High-resolution (10-30 m) brook trout surveys have been conducted annually since 1997 on the Staunton River in Shenandoah National Park VA, a high-gradient, boulder-dominated, headwater stream. While traditional weighted usable area (WUA) calculations for each habitat complex (10-30 m subreach) based on the model results had a positive correlation with young of the year (YOY) brook trout populations, there was no relationship between WUA and age 1+ brook trout. Spatial clusters of low velocity (LV), high depth (HD), low Froude number (LFr), and high WUA (HWUA) were identified and compared to age 1+ brook trout biomass in each habitat complex. There was no significant relationship between brook trout populations and HD; however LFr, LV and HWUA were highly correlated to each other and were positively correlated to brook trout biomass. Individual habitat patches identified as spatial low velocity outliers were also analyzed to calculate kinetic energy gradients (KEG) and spatial distribution of these potential feeding areas; however, there was no significant relationship between these areas and brook trout biomass.

3.2. Introduction

Two-dimensional (2D) depth-averaged hydrodynamic models for habitat evaluation applications have come into widespread use due to the ability to model spatial variation of velocity vectors in 2D (Lacey and Millar 2004; Hayes et al. 2007; Clark et al. 2008; Lee et al. 2010; Waddle 2010). The use of 2D models has implications for fish habitat modeling, because 2D models potentially allow the modeling of meso-scale habitat features such as wakes behind boulders (Waddle et al. 2000; Crowder and Diplas 2006). For example, many riverine salmonid species utilize flow obstructions as velocity shelters to minimize energy expenditure (Bjornn and Reiser 1991). While flow patterns around protruding boulders may be largely 2D (Tritico and Hotchkiss 2005), the ability of 2D models to represent these complex flows is uncertain. Because of difficulty in accurately measuring both velocity and topography, 2D hydraulic models typically yield predictions within 20-30% of measured velocity and are limited to channel types adhering to model assumptions (Pasternack et al. 2006). When boulders (or other flow obstructions) are submerged, 2D models are unable to represent 3D flow refugia that may be important to fish (Shen and Diplas 2008). In addition, while some 2D hydrodynamic models such as River 2D (Steffler and Blackburn 2002) have unsteady flow components, these models are unable to calculate turbulence parameters of flow structures utilized by swimming fish (Liao et al. 2003; Tritico and Hotchkiss 2005; Smith et al. 2006; Harvey and Clifford 2009; Roy et al. 2010). Despite these limitations, 2D hydraulic models are being used to predict and characterize habitat often utilizing habitat suitability indices (HSI) and weighted-usable area (WUA) to quantify habitat (Lacey and Millar 2004; Stewart et al. 2005; de Jalon and Gortazar 2007; Waddle 2010).

There are several hydraulic models that have been used to simulate hydraulic habitat suitability. The Physical Habitat Simulation Model (PHABSIM) is a well-established hydro-ecological model that provides a suite of tools for the numerical modeling of hydraulic habitat suitability for fish and invertebrate species based on field measurements of channel slope, water depth, velocity, and substrate (Bovee, 1982; Maddock, 1999; Booker and Dunbar, 2004); however, PHABSIM is one-dimensional (1D) and averages velocity, substrate, and depth values between measured cross sections and cannot account for the range of habitat types or physical

conditions adjacent to an individual fish location within a stream (Maddock, 1999). Similar to PHABSIM, 2- and 3D hydraulic models, can be combined with HSI for target species to predict the weighted usable area WUA, or relative amount of preferred habitat available to a target species (Ghanem et al. 1996; Booker 2003; Lacey and Millar 2004). However, WUA ignores local flow complexity and predicts habitat suitability based on species-specific flow and depth requirements. These measures ignore the spatial distribution of WUA, which may provide a more realistic view of habitat within the stream (Clark et al. 2008). The advantage of 2D and 3D habitat models over the conventional 1D model PHABSIM is the ability to spatially determine depth, and depth-averaged velocity and flow direction; thereby allowing the user to evaluate areas of particular ecological importance such as refugia from high flows.

It is difficult to describe habitat in headwater streams dominated by boulders in terms of longitudinal classifications such as riffles and pools (i.e. Gordon et al. 2004) because at any one cross section, the channel may contain a complex morphology that includes both areas of deep, slow-moving water and shallow fast-moving water with coarser bed materials. Although these streams tend to be dominated by a step-pool sequence longitudinally, the distribution of flow refugia (behind boulders, in local pool areas or local edge habitat) is 2D or even 3D. While pools may be easy distinguish at low flows, biologically important areas such as wakes behind boulders may be missed. There are a number of proposed methods for classifying habitat units including hierarchical clustering of velocity (Ward's algorithm; Emery et al. 2003), fuzzy logic (Legleiter and Goodchild 2005), Froude number (Fr, Eq. 1; Jowett 1993; Kemp et al. 2000), or expert-based (Moir and Pasternack 2008). Fr can be expressed as:

$$Fr = \frac{V}{\sqrt{gh}} \qquad (1)$$

where V is the depth-averaged velocity (m/s), g is the gravitational acceleration constant (9.81 m/s^2) and h is the flow depth (m). These methods using Fr to identify physical habitat classify habitat based on point measurements over the entire distribution of measurements and do not compare measurements to adjacent measurements. An alternate method of clustering is local indicators of spatial association (LISA) which are disaggregate measures of autocorrelation that

describe the extent to which particular areal units are similar to or different from their neighbors (O'Sullivan and Unwin 2003). For example, the local form of Moran's I provided in ArcGIS (ESRI, Redland, CA) can indicate whether the apparent similarity (spatial clustering of either high or low values) or dissimilarity (spatial outlier) are more pronounced than would be expected in a random distribution (ESRI 2010). If analyzing velocity distributions, for example, the spatial outliers can indicate areas of low flow surrounded by areas of high flow that could be used by feeding salmonids. Once patches are defined, spatial relationships can be further investigated using landcape ecology metrics to determine spatial patchiness, diversity, or connectivity (Schlosser 1991; Turner et al. 2001).

Stronger spatial changes in kinetic energy would be found in complex flows such as flows around an obstruction (boulder). Identifying and quantifying the relative habitat area of specific recirculation zones (wakes behind boulders) may be used to describe salmonid feeding locations where fish rest in relatively slow-moving water that is adjacent to faster water that transports food (Crowder and Diplas, 2006). Crowder and Diplas (2006) KEG predict ideal feeding habitat for brook trout with KEG rangng from 4 to 14 m^{-1}.

$$\left| \frac{\partial}{\partial x} \frac{V^2}{2} \right| = \left| \frac{2V_{AVE} \dfrac{V_2 - V_1}{\Delta x}}{V_1^2} \right| = KEG \qquad (2)$$

The overall goal of this research was to assess the characteristics of in-stream habitat that need to be considered for successful stream restoration. The specific objectives of this study were to 1) quantify in-stream habitat based on 2D model results, 2) to examine the influence of topography on modeling results, and 3) to examine the spatial distribution of individual habitat patches within a complex boulder-dominated headwater stream.

3.3. *Methods*

3.3.1. Study Site

The Staunton River is a second-order headwater stream originating on the eastern slope of the Blue Ridge Mountains in Shenandoah National Park (SNP), VA (Figure 3-1). This high-gradient (10% slope) channel has an average width of 3.5 m, consists of pools separated by step-pool cascades, small (<2 m) waterfalls, and bedrock slides, and drains 11 km^2 (Hyer et al. 1995). The 100 m study reach was a 7 % slope single channel containing six habitat complexes, no major tributaries, and no major waterfalls or wood jams. Long-term brook trout *Salvelinus fontinalis* habitat monitoring has been conducted along the Staunton River following an 100 – yr flood and resulting debris flow in June 1995 (Roghair 2005). The Shenandoah Watershed Study (SWAS; Department of Environmental Sciences at the University of Virginia, Charlottesville) has maintained a flow and water-quality monitoring station near the confluence with the Rapidan River since 1993 (Figure 3-1). The 100 m reach used for this study is approximately 2.5 km upstream of this confluence and water discharge rates in the study reach were calculated by adjusting the gaged flow rate by the watershed size upstream of the study reach outlet (7.3 m^2) using drainage-area ratios (Reis III 2007). High flows during the study period ranged up to 7 m^3/s (Figure 3-2). In general, the highest median flows occurred in December and the lowest flows occurred in August (Figure 3-3).
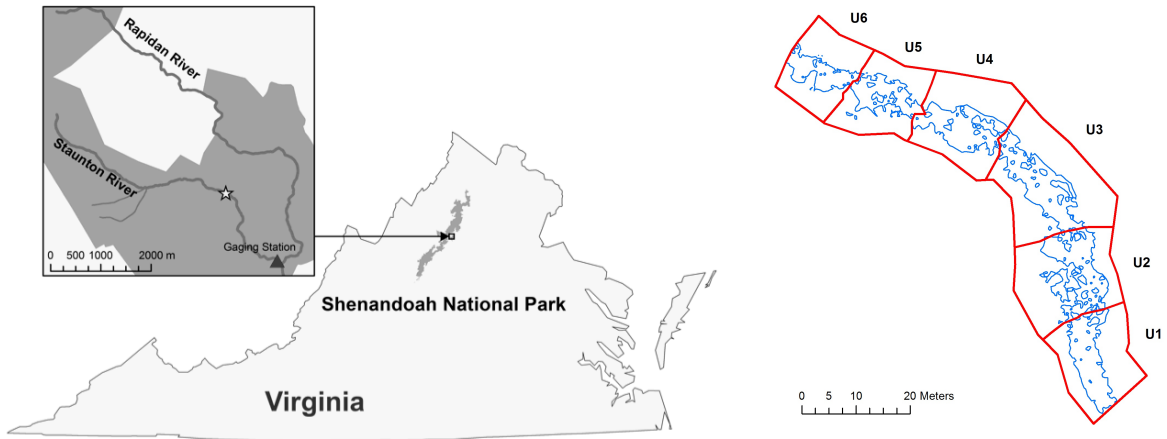
Figure 3-1. The Staunton River is in Shenandoah National Park, Virginia, USA. The study reach (star) was divided into six habitat complexes.
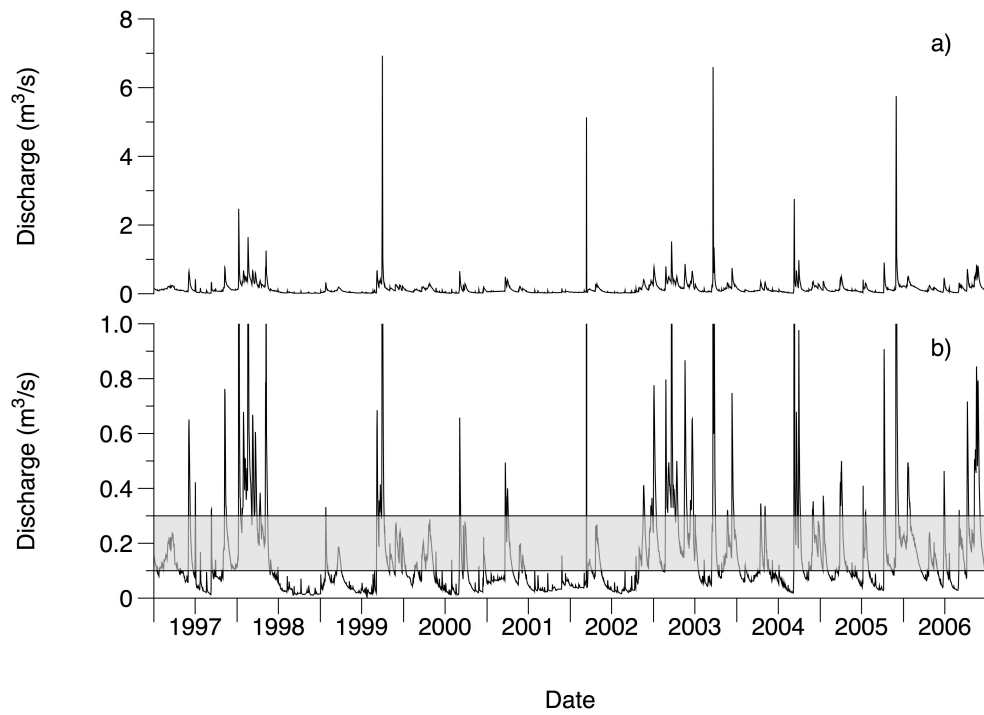
Figure 3-2. Staunton River discharge over the study period (1997-2006) for a) entire range of flow rates and b) enlarged view of flow rates under 1 m³/s. Shaded area is the range of modeled flows.
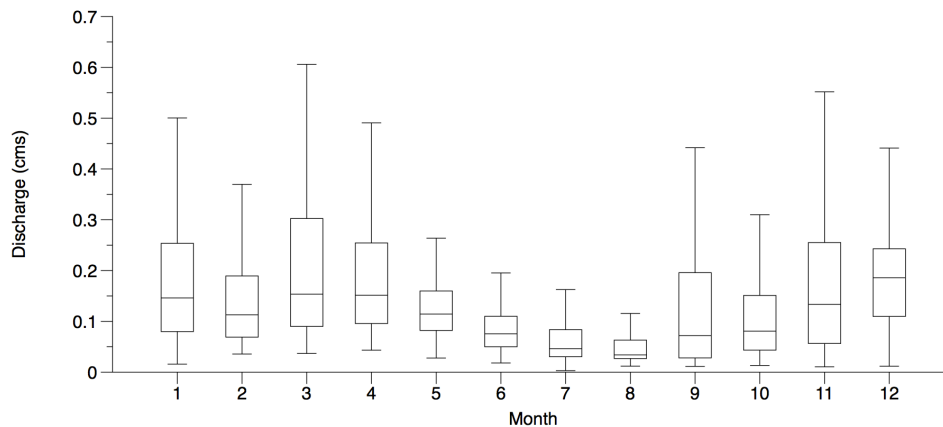
Figure 3-3. Discharge in the Staunton River by month over the study period (1997-2006). Boxes represent the median value bounded by the 25th and 75th percentile. Whisker are 1.5 times the inter quartile range.

The brook trout population data used in this study were described in detail in Kozarek et al. (2010); this study focused on only the upper reach described. The brook trout abundance data were collected as part of a recolonization and post-recolonization brook trout movement study (Roghair et al. 2002; Roghair 2005) following a debris flow in 1995 that eliminated a 30-m band of the riparian area in the lower 1.9 km of the Staunton River (Karish et al. 1997). The subset of these data used in this study ranged from 1997 to 2004 in autumn (mid-October) and from 1997-2007 in spring (mid-May to mid-June) in a 100-m reach approximately 0.6 km upstream of the debris flow affected area. Discharge measurements corresponding to the brook trout sampling dates ranged from baseflow (0.02 m^3/s) to 0.3 m^3/s. Fish were located by capture using a backpack electrofishing unit (single pass; 73% first pass capture efficiency; Roghair 2005) by subdividing the river into 10-30-m habitat complexes comprised of multiple pools and riffles and terminating at potential low flow barriers. The length (mm), weight (g), and location of each fish were recorded for each fish (Roghair 2005); age 1+ ($\geq$ 100 mm) and young of the year (<100 mm) populations were separated.

3.3.2. 2D Hydraulic Modeling

We examined the influence of topography measurement on 2D hydraulic model (River 2D; Steffler and Blackburn 2002) accuracy in complex channels. Detailed modeling methods were described in Kozarek et al. (2010). The model was calibrated to measured water surface, depth, and velocity by adjusting the roughness, k_s, and eddy viscosity, ε.

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To examine the influence of the method of topographic data collection on model results, two different methods were used to measure topography for this study, traditional total station surveying (Topcon GTS-230W, Livermore, CA) as described in Kozarek et al. (2010) and terrestrial laser scanning (TLS; Optech ILRIS-3D, Belgrade, MT). A detailed comparison of the two survey methods for this study site is described in Resop (2010). To represent the channel topography, the total station survey (>1 point every 0.5 m^2) had a greater density of points in complex areas. Boulders were represented by taking a point at the apex(es) of each boulder and points around the base. The TLS survey consisted of 89 total scans at 25 stations along both sides of the channel to capture all sides of protruding boulders. These scans were aligned in PolyWorks IMAlign (InnovMetric Software Inc., Quebec City, CA; InnovMetric 2008) by first using the N-point alignment function, manually checking that the scans were aligning correctly, and then using the best-fit alignment and comparison algorithm. After all scans were aligned the total number of points was reduced by first manually selecting and removing large vegetation along the banks in IMInspect. The TLS scan cannot capture points beneath the water surface, therefore, the TLS survey was aligned with the total station survey in IMAlign using six benchmarks in the total station survey that were marked in the TLS survey.

We used two different merging methods because combining the two surfaces automatically creates a bias toward the TLS topography due to the difference in point density between TLS (approximately 1 point per cm) and total station (approximately 1 point per 0.5 m^2). In the first method, the aligned topographic models were merged in IMMerge and compressed in IMEdit to reduce the final number of vertices. The second method utilized the 2-cm digital elevation model (DEM) described in Resop et al. (2010) created from the TLS data and a separate 2-cm DEM created from the total station survey data in ArcGIS (ESRI, Redland, CA) using a natural neighbor interpolation method. These DEMs were overlaid and run through a single pass filter to remove edge errors from the TLS. The model results from each topographic model (total station, TIN, and DEM) were compared to each other to investigate the importance of topography on 2D model results (Table 3-1).

Table 3-1. Comparison of modeled depth, velocity, and water surface area in each habitat complex for three different methods of deriving study reach topography. Percent differences are the differences between the TLS derived topography and the total station topography model results.

Total Station	Depth (m)				Velocity (m/s)		Area (m^2)	
	Maximum	Difference	Mean	Difference	Mean	Difference	Total	Difference
U1	0.40	-	0.20	-	0.29	-	80.8	-
U2	0.56	-	0.18	-	0.23	-	96.8	-
U3	0.52	-	0.19	-	0.29	-	96.6	-
U4	0.59	-	0.19	-	0.28	-	80.1	-
U5	0.99	-	0.24	-	0.30	-	54.4	-
U6	0.63	-	0.23	-	0.28	-	47.5	-
DEM								
U1	0.50	25%	0.22	13%	0.29	-1%	78.4	-3%
U2	0.44	-21%	0.13	-26%	0.33	39%	91.9	-5%
U3	0.44	-16%	0.16	-18%	0.38	33%	81.2	-16%
U4	0.50	-16%	0.16	-15%	0.37	33%	75.4	-6%
U5	0.94	-5%	0.20	-16%	0.38	25%	54.6	0%
U6	0.49	-21%	0.18	-20%	0.38	33%	45.4	-4%
TIN								
U1	0.40	0%	0.19	-3%	0.27	-6%	85.6	6%
U2	0.59	5%	0.16	-10%	0.24	3%	104.3	8%
U3	0.46	-11%	0.19	-2%	0.29	2%	95.1	-2%
U4	0.67	12%	0.20	8%	0.27	-1%	80.2	0%
U5	0.88	-11%	0.20	-16%	0.33	10%	53.3	-2%
U6	0.65	4%	0.23	-1%	0.29	4%	48.7	2%

Although TLS data were available, we did not use the model results based on the TLS derived topography because incorporating the TLS data into the modeled topography did not improve the fit between modeled and measured depth and velocity compared to the results from Kozarek et al. (2010). Both methods used to incorporate TLS data into model topography resulted in large differences in average habitat complex model results, depth (up to 26 %) and velocity (up to 39%). The difference in wetted area for each topographic model scenario was much smaller (maximum difference of 16%). It is possible that with a better method of incorporation of TLS data or with further calibration that model results could improve.

3.3.3. Weighted Usable Area

Physical habitat was calculated using habitat suitability indices described in Baker and Coon (1997) using the River 2D habitat capabilities (Steffler and Blackburn 2002) that calculate WUA using equation 3.

$$WUA = \sum s_v s_d s_{ci} a_i \qquad (3)$$

where s_v is the habitat suitability index for velocity, s_d is the habitat suitability index. for depth, s_{ci} is the habitat suitability index for channel index and a_i is the area of node i. The combined suitability index (CSI) for a location is the WUA without incorporating area. The WUA was calculated for each habitat complex at 0.1 m^3/s, 0.2 m^3/s, and 0.3 m^3/s.

3.3.4. Patch Classification

Model results for Fr, depth, velocity, and CSI were exported on a 0.05-m grid for the 0.3 m^3/s model. ArcGIS (ESRI 2010) was used to calculate spatial clusters and outliers of depth, Fr, velocity and CSI using the Cluster and Outlier analysis tool. This tool uses the Anselin local Moran's I method of identifying spatial clusters with attribute values similar in magnitude and spatial outliers. Each point is classified as a spatial clusters of high (HH) and low (LL) values; spatial outliers of high values surrounded by low values (LH) and low values surrounded by high values (HL); and values that are not significantly spatially related (NS) (Anselin 1995; ESRI 2010). These clusters were used to define habitat patches of high depth (HHD), low Fr (LLFr), low velocity (LLV), and high CSI (HHCSI) to represent preferred age 1 + brook trout habitat patches including deep low velocity locations described by Sotiropoulos et al. (2006). In addition, spatial outliers of low velocity surrounded by high velocity (HLV) were identified and quantified to represent possible feeding locations as described by Crowder and Diplas (2006). The total area of each of these patches was compared to the age 1+ brook trout biomass. In addition, the spatial distribution of the low velocity outliers was investigated using FRAGSTATS (McGarigal et al. 2002) to calculate the *proximity* and *aggradation index* of low velocity outliers in each habitat complex. The KEG across each identified low velocity outlier

area was calculated using the minimum velocity within the patch and the maximum velocity within 0.05 m of the patch boundary.

3.3.5. Statistical Analysis

Statistical evaluations of brook trout population data were completed using JMP software (SAS Institute Software 2007). Comparisons of brook trout populations between seasons were conducted using the non-parametric Mann-Whitney Rank Sum test ($\alpha = 0.05$) (Conover 1999); comparisons of more than two groups were conducted using the Kruskal-Wallis nonparametric one-way analysis of variance ($\alpha = 0.05$) (Ott and Longnecker 2001). Non-parametric Spearman's correlation analysis was used to relate habitat variables with brook trout abundance.

3.4. Results

3.4.1. Brook Trout Population

Analysis of brook trout populations indicated that while number and mass of fish differed between habitat complex, there was no effect of season on the number and mass of brook trout found in each habitat complex. The brook trout data were split into age 1+ and young of the year (YOY; less than 100 mm) populations. There were no significant differences (Mann-Whitney Rank Sum test; $\alpha = 0.05$) between number of fish normalized by habitat complex length or BTM per habitat complex length for either population by season; therefore, the data were combined for spring and fall.

There were significant differences in both brook trout abundance and BTM between habitat complexes (Table 3-2; Kruskal-Wallis nonparametric one-way analysis of variance; $\alpha = 0.05$). Brook trout abundance and BTM were highest in complex U2 for both age 1+ and YOY populations, and lowest in complex U3 for age 1+ populations. While complex U6 had the second lowest total brook trout abundance, the density of brook trout was the third highest indicating that while there was a smaller number of fish overall, this complex supported larger fish.

Table 3-2. Fish abundance and density in each habitat complex for all sampling dates and young of year (YOY; < 100 mm) fish abundance and density for spring sampling dates.

Habitat Complex	Length (m)	Total Fish Density (# Fish/m)		Total Mass Density (g/m)		Age 1+ Fish Density (# Fish/m)		Age 1+ Mass Density (g/m)		YOY Fish Density (#Fish/m)		YOY Mass Density (g/m)	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
U1	17	$1.01^{a,b}$	0.59	$15.35^{a,b}$	11.25	0.40^{b}	0.31	$13.55^{a,b}$	11.16	$0.61^{a,b}$	0.48	$1.80^{a,b}$	1.80
U2	18	1.43^{a}	0.73	22.83^{a}	9.33	0.72^{a}	0.33	21.63^{a}	9.39	0.79^{a}	0.73	2.54^{a}	2.91
U3	20	0.67^{b}	0.28	9.75^{b}	5.35	0.40^{b}	0.26	10.15^{b}	6.07	$0.39^{b,c}$	0.24	$1.31^{a,b}$	0.96
U4	15	$1.07^{a,b}$	0.49	20.01^{a}	11.35	$0.52^{a,b}$	0.31	$16.18^{a,b}$	9.78	$0.43^{a,b,c}$	0.30	$1.47^{a,b}$	1.25
U5	15	0.84^{b}	0.39	$15.05^{a,b}$	6.45	0.43^{b}	0.22	12.23^{b}	5.68	$0.31^{b,c}$	0.27	$1.06^{a,b}$	0.94
U6	13	0.78^{b}	0.38	$16.56^{a,b}$	11.02	0.40^{b}	0.23	11.95^{b}	8.31	0.20^{c}	0.15	0.71^{b}	0.67

Levels not connected by same letter are significantly different (Tukey-Kramer HSD)

3.4.2. Weighted Usable Area

To compare the effect of topography on WUA calculations, WUA for YOY and age 1+ brook trout populations was calculated for each habitat complex using the 0.3 m^3/s model results from each topography scenario (Table 3-3). While local topography is very important in predicting local velocity and depth characteristics (Waddle 2010), when using the WUA model results, the differences between model results with different topography were small between the total station and TIN topography, but ranged up to 51% for the DEM topography. The DEM topographic model predicted shallower faster flows on average in each habitat complex (Table 3-1), which had a drastic impact on WUA calculations.

Table 3-3. Comparison of modeled WUA in each habitat complex for three different methods of deriving study reach topography. Percent differences are the differences between the TLS derived topography and the total station topography model results.

Total Station	YOY			Age 1+		
	WUA (m^2)	Difference	% total water surface area	WUA (m^2)	Difference	% total water surface area
U1	48.2	-	60%	30.8	-	38%
U2	51.5	-	53%	32.5	-	34%
U3	49.3	-	51%	32.6	-	34%
U4	43.2	-	54%	26.4	-	33%
U5	23.1	-	42%	19.0	-	35%
U6	25.7	-	54%	20.2	-	43%
DEM						
U1	44.5	-8%	57%	37.7	22%	48%
U2	35.1	-32%	38%	18.2	-44%	20%
U3	28.6	-42%	35%	16.0	-51%	20%
U4	27.1	-37%	36%	16.7	-37%	22%
U5	17.4	-25%	32%	14.2	-25%	26%
U6	18.7	-27%	41%	12.3	-39%	27%
TIN						
U1	48.9	2%	57%	30.8	0%	36%
U2	56.4	10%	54%	30.4	-6%	29%
U3	50.0	1%	53%	31.9	-2%	34%
U4	42.7	-1%	53%	29.6	12%	37%
U5	22.5	-2%	42%	16.0	-16%	30%
U6	27.3	6%	56%	22.1	9%	45%

For all three topography scenarios, the YOY WUA was significantly (but slightly) correlated to the YOY BTM in each habitat complex. The differences between topographic model results were greater for the age 1+ brook trout populations, however, and only the WUA results for the DEM model were significantly correlated to the age 1+ BTM. These results indicate that that despite modeling errors due to topography, which can be large when looking at individual measurement points (e.g. Kozarek et al. 2010 and Waddle 2010), WUA may be a predictor of fish mass for individual life stages (e.g. YOY) within the same reach and when averaged over a sub-reach (10 – 30 m). However, for age 1+, WUA modeled in each habitat complex was not correlated to BTM for two topography cases and weakly correlated in the case that provided the most different depth and velocity model results (DEM). For habitat complex U3 (low age 1+ BTM) had high WUA while habitat complex U2 (high age 1+ BTM) had low WUA indicating that the WUA method using these HSI preference curves does not adequately capture age 1+ brook trout habitat. YOY brook trout have narrower preference ranges for both

depth and velocity and prefer shallow, slower flows than older fish (Baker and Coon 1997). WUA was calculated at each model node and was summed for the habitat complex; therefore, the WUA at each point cannot represent the spatial distribution of depth or velocity and may misrepresent important habitat patches such as feeding locations in wake areas behind boulders adjacent to faster moving water (Crowder and Diplas 2006).

Table 3-4. Spearmans's Rank coefficients (ρ) and *p*-values comparing BTM to WUA calculated at 0.3 m^3/s. in each habitat complex for three different topography scenarios.

	YOY		Age 1+	
Topographic Model	ρ	*p*-value	ρ	*p*-value
Total Station	0.2908	0.0029	0.0666	0.4999
DEM	0.3117	0.0013	0.1926	0.0491
TIN	0.2908	0.0029	-0.0293	0.7663

The influence of discharge on modeled (total station topography) WUA calculations was investigated by comparing WUA for YOY and age 1+ brook trout at 0.1 m^3/s, 0.2 m^3/s, and 0.3 m^3/s (Figure 3-4). These results indicate that there was no relationship between the calculated age 1+ WUA and BTM at all three flow rates. The age 1+ WUA in habitat complex U3 was high at all three flow rates, while the BTM measured in U3 was the lowest of all habitat complexes corresponded to the highest and lowest fish populations. These results also indicate that the model performed best at winter flow rates (approximately 0.3 m^3/s) for WUA calculated for YOY.
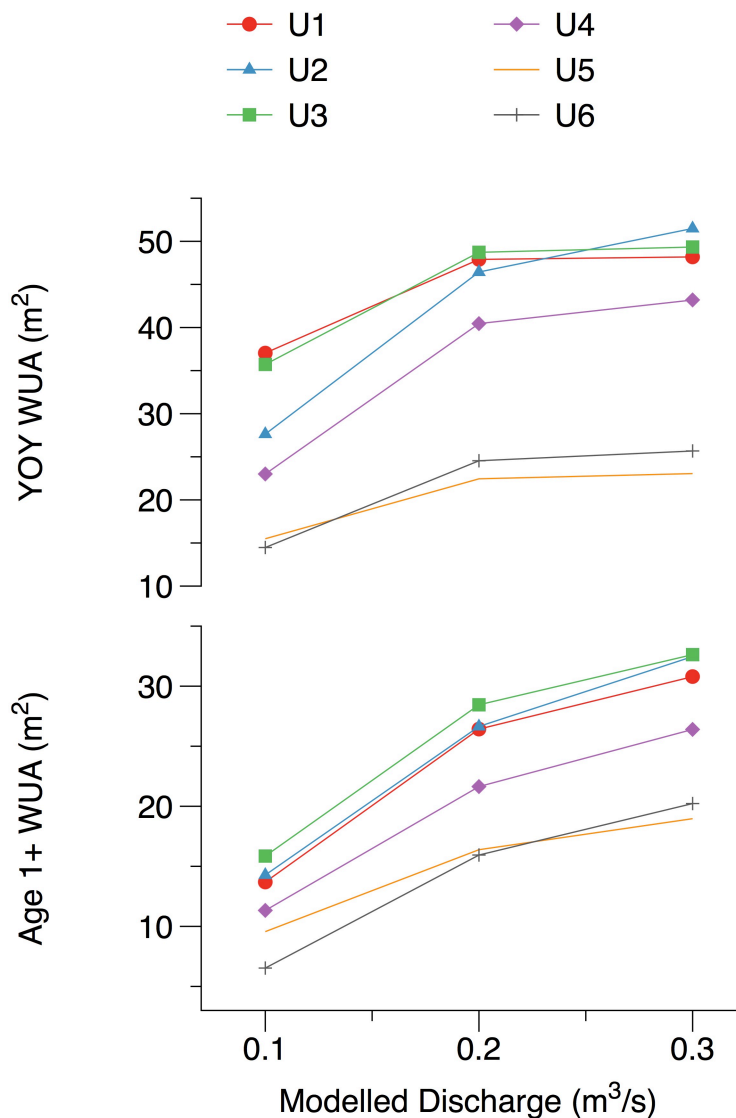
Figure 3-4. Calculated WUA for age 1+ and young of year (YOY) from total station –derived topography model results.

3.4.3. Spatial Distribution of Habitat Patches

Local patches of statistically similar ($\alpha = 0.05$) spatial clusters of depth (D), velocity (V), Fr, and Age 1+ CSI were calculated from the total station model results using the local form of Moran's I (ESRI 2010). This method classified each modeled point (0.05-m grid) as a high value cluster (HH), a low value cluster (LL), a high value spatial outlier (HL), a low value spatial

outlier (LH), or not significantly clustered. For example, small areas of relatively high CSI or transitional areas of CSI were not classified as HHCSI (Figure 3-5). Figure 3-6 illustrates the HHD and LLD patches over the modeled water's edge. Habitat complexes U2, U5, and U6, had large deep areas just downstream of the habitat complex boundary (low flow barrier), while U1, U3, and U4 had deep areas that were located more mid-stream. Over the entire reach, HHD values ranged 0.27-0.99 m with a mean of 2.93 m. The mean maximum depth of individual patches was 0.47 m. LLV, LLFr, and HHCSI patches were similarly distributed (Figure 3-7 and Figure 3-8). The HHV values ranged from 0.32-1.28 m/s and the LLV values ranged from 0.0043-0.30 m/s. The LLFr values ranged from 0.002-0.19 (similar to the classification of pool by Jowett 1993 for Fr less than 0.19), while HHFr values ranged from 0.21-0.99 and spanned the classification runs and riffles by Jowett (1993).   In general, HHCSI patches showed good agreement with LLFr patches with the exception of habitat complex U1 (Figure 3-8).   Because age 1+ brook trout prefer high depth, low velocity patches or pools (Sotiropoulos et al. 2006) or feeding areas of low velocity located next to high velocity (Crowder and Diplas 2006), data analysis focused on HHD, LLV, LLFr, HHCSI, and LHV. Table 3-5 is a summary of the number, mean area, and total area of HHD, LLV, LLFr, and HHCSI patches within each habitat complex. These patches were defined at a finer spatial scale than the collected fish data and represent patches at a single steady state flow condition. Individual fish location surveys will need to be conducted to identify whether these areas are occupied at this flow.
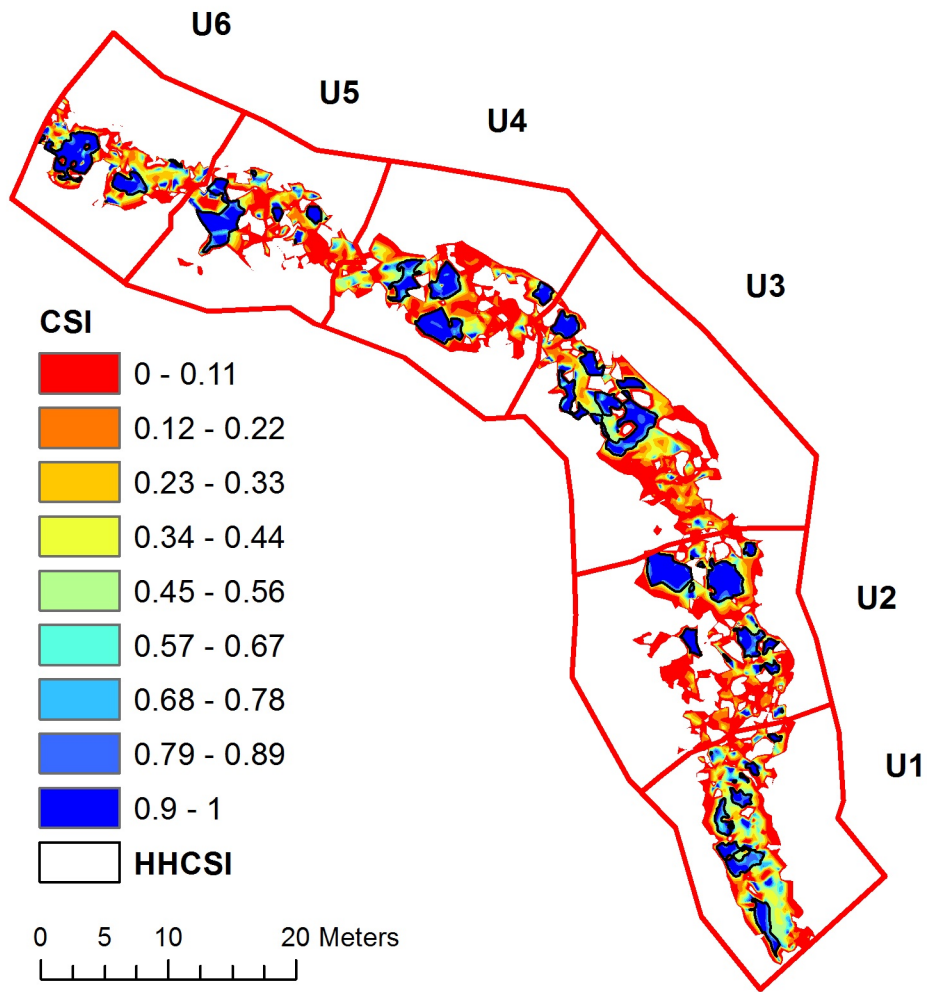
Figure 3-5. Combined suitability index model results for 0.3 m³/s flow rate overlaid by HHCSI (high combined suitability index) patches. Areas not classified as HHCSI were DDCSI, HLCSI or LHCSI outliers, or not significantly clustered.
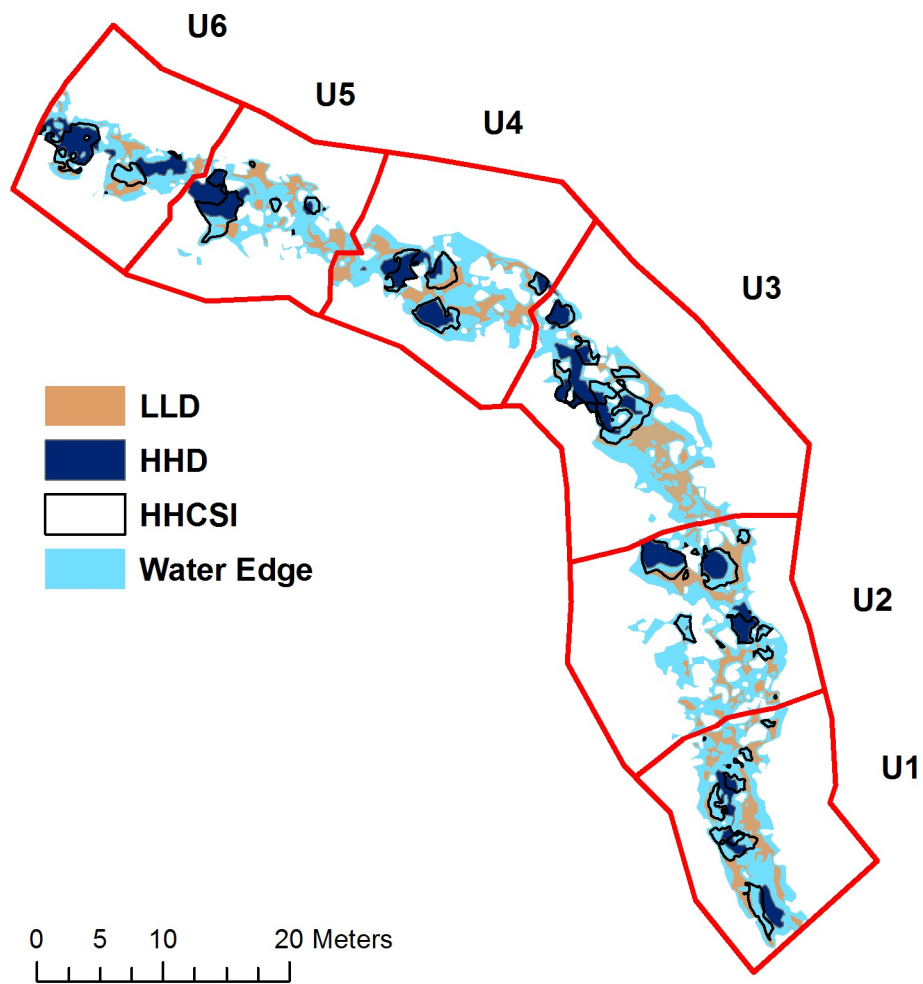
Figure 3-6. HHD (high depth) and LLD (low depth) patches for the 0.3 m^3/s model results over the modeled water's edge. Areas not classified as HHD or LLD were either HLD or LHD outliers or not significantly clustered.
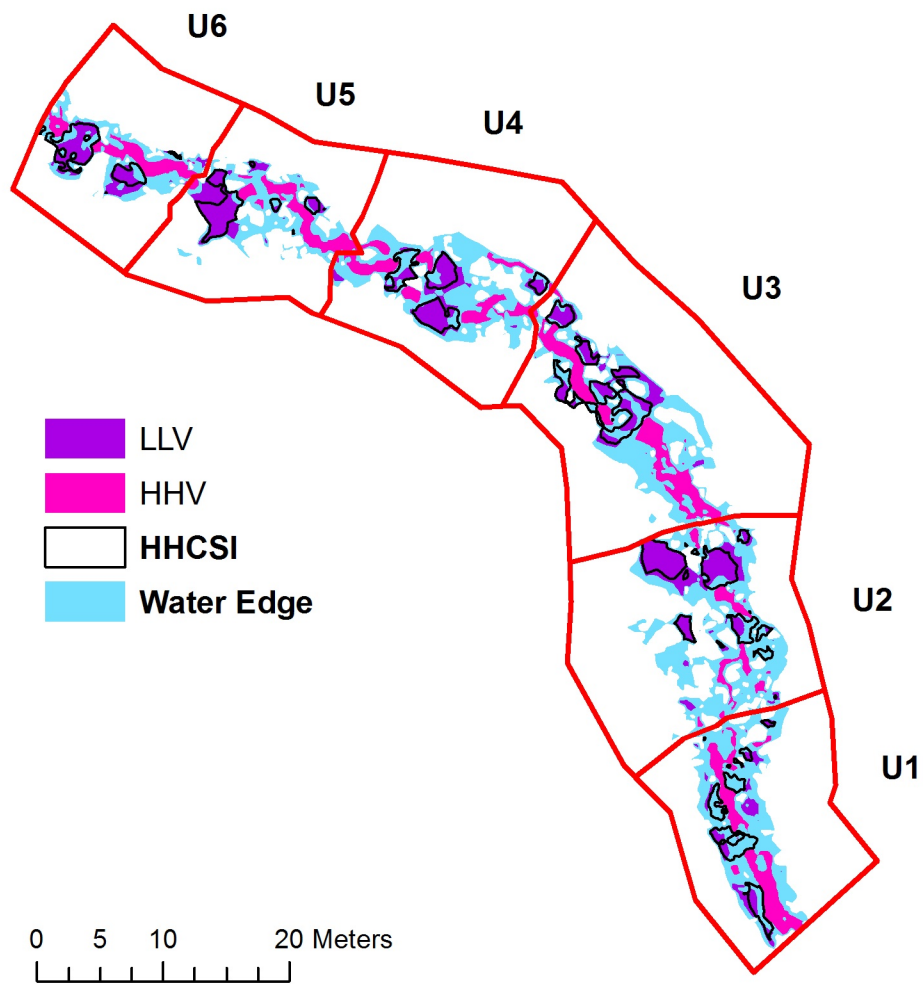
Figure 3-7. HHV (high velocity) and LLV (low velocity) patches for the 0.3 m³/s model results over the modeled water's edge. Areas not classified as HHV, LLV, or LHV were either HLV or not significantly clustered.
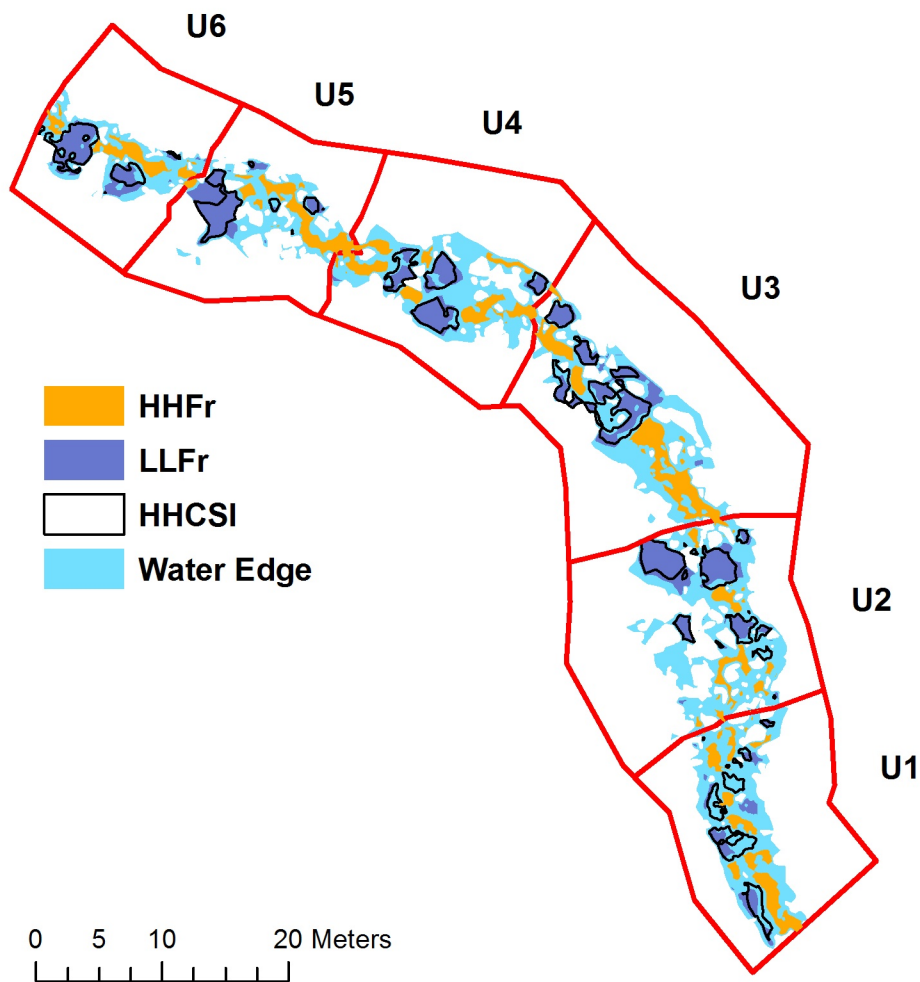
Figure 3-8. HHFr and LLFr patches calculated from the 0.3 m³/s model results. Black lines indicate HHCSI patches over the modeled water's edge. Areas not classified as HHFr or LLFr were either HLFr or LHFr outliers or not significantly clustered.

A non-parametric correlation analysis between age 1+ BTM and the area of HHD, LLV, LLFr, HHCSI and LHV patches indicates that for age 1+ brook trout, the LLV, LLFr, and HHCSI patches within each habitat complex were weakly ($\rho \approx 0.2$) but significantly correlated to the age 1+ BTM ($\alpha = 0.05$) and strongly correlated to each other (Table 3-6). Although brook trout preferred high depth, low velocity areas in previous studies (Sotiropoulos et al. 2006), there was no relationship between HHD patch area and age 1+ BTM in this study. There was a slight negative correlation between BTM and LHV patch area ($\rho = -0.2384$; *p-value* = 0.0143) and a

slight negative correlation between LLV and LHV patch area ($\rho = -0.8286$, *p-value* < 0.0001) indicating that larger low velocity shelters (LLV) have a greater relationship to age 1+ brook trout populations than smaller more distributed velocity shelters (LHV). However, these smaller shelters may still play an important role in habitat quality, but further research is need to identify where fish are located and which areas are most often occupied by fish.

Table 3-5. Area of identified habitat patches in each habitat complex.

Habitat	HHD Patch Area			LLV Patch Area			HHCSI Patch Area			LLFr Patch Area		
	#	Mean	Total	#	Mean	Total	#	Mean	Total	#	Mean	Total
U1	6	1.4	8.4	13	0.7	8.9	9	1.4	12.7	12	0.7	8.5
U2	5	2.5	12.4	14	1.8	25.4	13	1.6	21.4	13	1.9	24.7
U3	4	3.7	14.9	13	1.4	18.1	7	2.8	19.3	13	1.6	21
U4	4	2.8	11.4	8	2.1	16.5	4	4.0	15.8	9	1.9	16.8
U5	3	3.6	10.7	8	1.9	15.3	6	1.7	10.2	7	2.3	15.9
U6	4	3.2	12.7	7	1.8	12.6	6	2.0	11.7	8	1.8	14.2

Table 3-6. Spearman's Rank correlation coefficients (ρ) and *p-values* comparing habitat patch areas to age 1+ BTM; *p-values* less than 0.05 indicate significant correlations.

Variable	ρ	*p-value*
LLV Patch Area	0.238	0.0145
LLFr Patch Area	0.238	0.0145
LHV Patch Area	-0.2384	0.0143
HHD Patch Area	-0.058	0.5565
HHCSI Patch Area	0.2263	0.0203

LHV patches were also calculated as a means to represent pockets of low velocity surrounded by high velocity such as would be expected in the wakes behind boulders. The spatial distribution and KEG across individual LHV patches was calculated (Figure 3-9; Table 3-7). Class-level landscape ecology metrics (*proximity* and *aggregation index*) were calculated for each habitat complex. The *proximity index* considers the size and proximity of all patches whose edges are within a specified search radius of the focal patch (McGarigal et al. 2002). A search radius equal to mean channel width, 3.5 m, was used. The aggregation index is calculated from an adjacency matrix, which shows the frequency with which different pairs of patch types (adjacencies between the same patch type) appear side-by-side on the map (McGarigal et al.

2002). These metrics describe the spatial distribution of the low velocity outlier habitat patches. For example, visually, habitat complex U1 has velocity shelters close to the main channel flow reflected in the proximity index. While these types of landscape ecology metrics could be important to describe the connectivity of individual habitat areas, in this case, the methods used to classify low velocity outliers classified very small areas that may not be biologically relevant, and missed large areas that would be expected to be wakes behind boulders (e.g. Figure 3-9, U2). With better modeling methodology to more accurately represent the complex velocity flow patterns around protruding boulders, more velocity shelters may be identified.

The mean KEG calculated for each low velocity outlier area in each habitat complex (Table 3-7) was much higher than the suggested range of KEG for brook trout feeding areas of 4-14 m^{-1} (Crowder and Diplas 2006). This is due to the fact that the lowest velocity within these areas was close to zero. The suggested range of KEG was calculated based on actual fish locations (Crowder and Diplas 2006) and not the minimum velocity within a velocity shelter. Therefore, KEG was calculated based on five cm gridded model output to examine spatial velocity gradients at the fish focal feeding length scale (Figure 3-10). While these small velocity shelters are not related directly to brook trout biomass, they could be important as feeding locations, but detailed surveys of individual brook trout locations would be needed to identify whether these sites are being utilized.
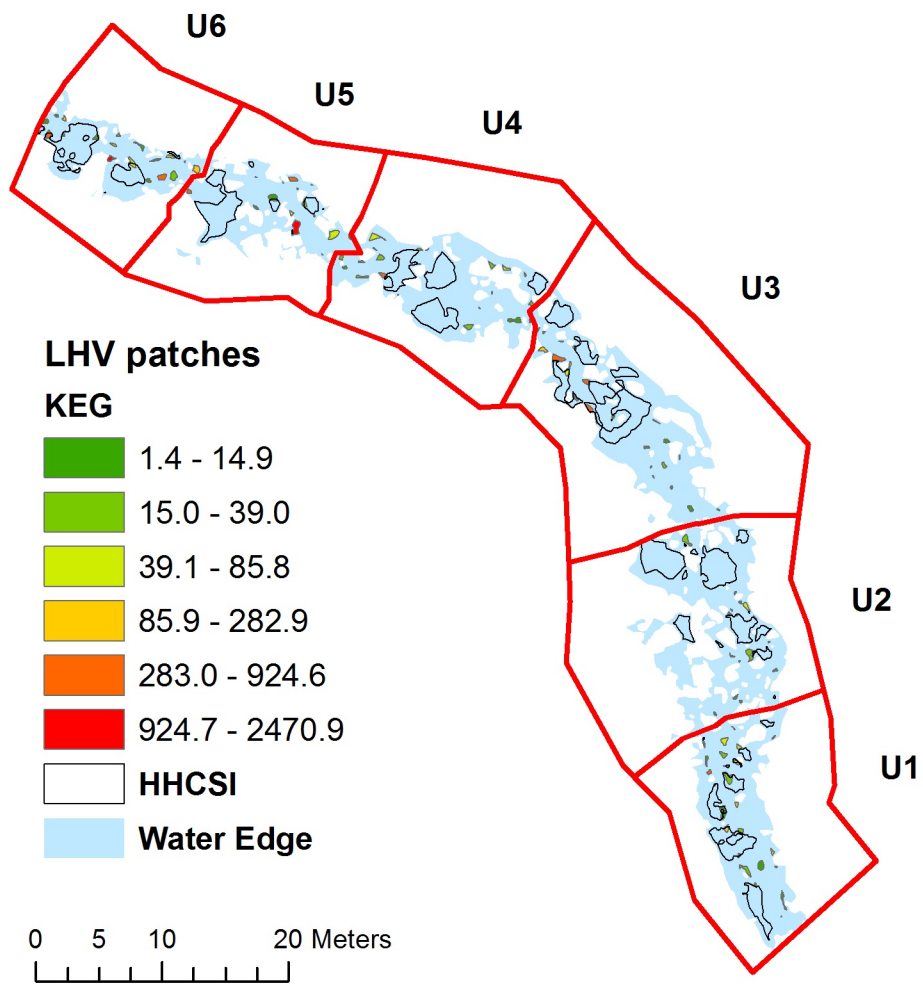
Figure 3-9. HLV (low velocity spatial outliers) patches scaled by KEG across each patch. Colors indicate strength of KEG for each patch, and dark blue area indicates area of high WUA.
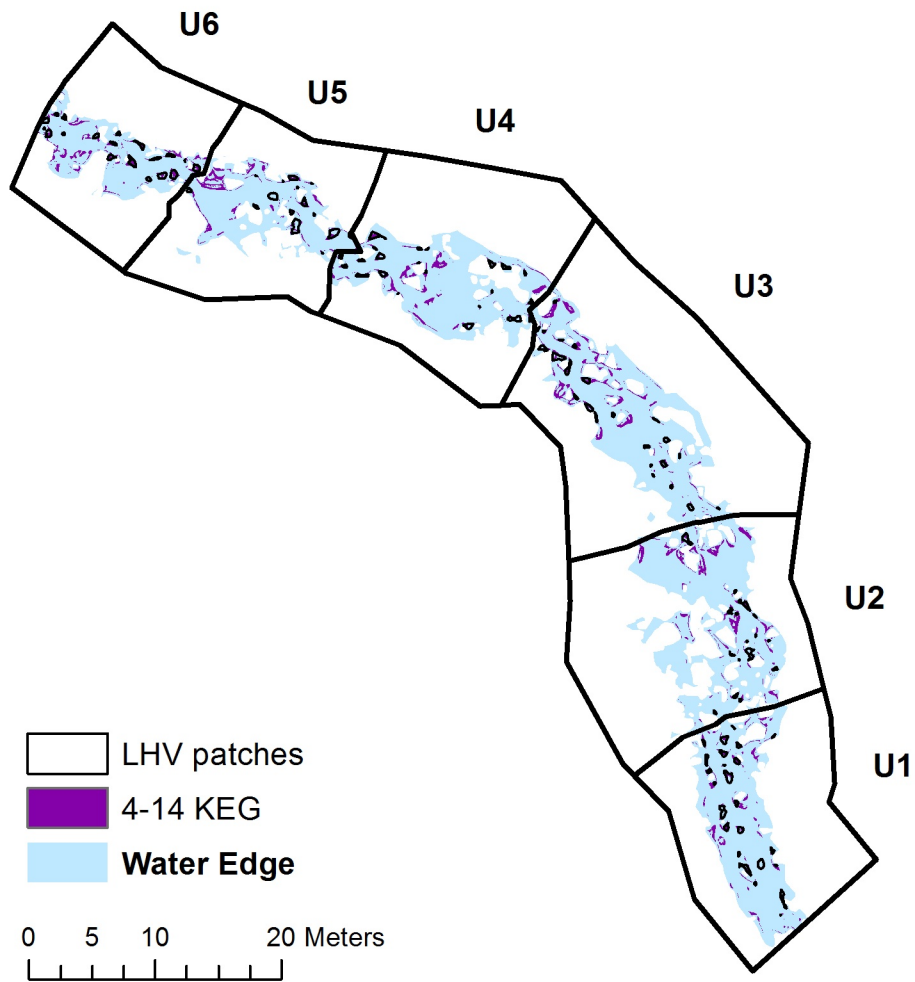
Figure 3-10. HLV (low velocity spatial outliers) patches. Purple color indicate areas of KEG between 4-14 m^{-1} as proposed by Crowder and Diplas 2006 as areas of brook trout feeding locations.

Table 3-7. Landscape ecology metrics and kinetic energy gradients (KEG) for LHV patches.

Habitat Complex	# LHV Patches	Proximity	Aggregation Index	KEG (m^{-1})			Area (m)	
				Min	Max	Mean	Mean	Total
U1	31	1.95	83.37	2.16	445.16	61.44	0.11	2.76
U2	30	1.61	77.63	2.80	924.57	67.85	0.06	1.07
U3	28	1.04	81.87	2.36	820.32	125.76	0.08	1.89
U4	28	1.52	81.82	1.40	603.31	69.03	0.08	1.93
U5	20	0.67	88.01	2.82	1809.10	209.06	0.12	1.88
U6	23	1.69	84.37	4.21	2470.87	228.19	0.12	2.28

While the TLS-derived topography was not used for the modeling portion of this study, the results of this study provide insight into the error involved in rapid habitat assessment methodology, specifically the visual habitat assessment conducted in Kozarek et al. 2010. Table 3-8 includes the results from the previous study (% pool, % riffle, and % protruding boulders) collected at low flow by rapid assessment compared to the % protruding boulders calculated using the TLS data (Resop 2010) and the % pool estimated from the area of LLFr patches in this study. There was no significant relationship between % pool and brook trout in Kozarek et al. 2010, but using the % pool estimated from the Fr, there was significant relationship. Similarly, if the age 1+ BTM (normalized by the habitat complex length) is used, there is no significant relationship between % boulder (visually estimated) and BTM (g/m). There is however, a stronger, but still weak, relationship between the % protruding boulders as calculated automatically from the TLS data and the age 1+ BTM ($\rho = 0.3286$ *p-value* = 0.0006; Figure 3-11). With a larger sample size (> 6 habitat complexes) this relationship may improve.

Table 3-8. Habitat variables for each habitat complex.

Habitat Complex	% Pool[a]	% Riffle[a]	% Boulders[a]	% Boulders (TLS)[b]	% Pool (Fr)[c]
U1	20	70	10	31.0	11
U2	40	40	44	42.3	26
U3	30	60	40	30.9	22
U4	15	35	55	34.1	21
U5	25	40	65	41.5	29
U6	40	45	30	24.2	30

[a]Kozarek et al. 2010
[b]Resop 2010
[c]estimated using 0.3 m^3/s model results
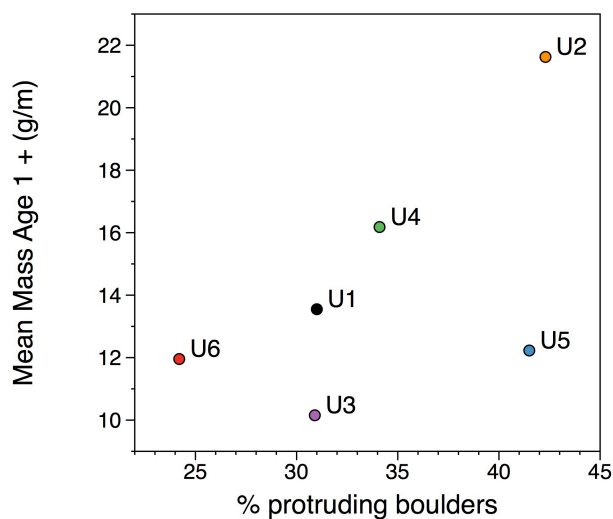
Figure 3-11. % protruding boulders calculated from TLS data (Resop 2010) versus length normalized age 1+ BTM.

3.5. Conclusions

Ultimately, while there was no relationship directly between summed WUA in each habitat complex and age 1+ BTM, when the spatial distribution of HSI was taken into account to quantify only clustered HHCSI habitat patches, WUA was as good a predictive variable as LLV or LLFr patches. This methodology identified areas of low velocity surrounded by high velocity that could potentially be feeding locations for age 1+ brook trout. These areas were negatively correlated with BTM, however, and alone do not describe habitat areas important for overall BTM. Because of the relationship between the % protruding boulders and BTM, it is likely that calculating only the influence of the low velocity areas ignores the influence of the complex flows around these obstructions. By default, a 2D hydraulic model cannot capture complex 3D flow which may be important to brook trout habitat (Shen and Diplas 2008) and cannot model turbulence characteristics important to salmonid habitat (Smith et al. 2006; Harvey and Clifford 2009).

Despite the widespread use of WUA metrics as a measure of habitat quality, this model adds to the literature suggesting caution when interpreting WUA results, especially when using non-site specific HSI curves (see review by Clifford et al. 2008). The clustering methodologies

used in this study suggest that the spatial relationship of WUA needs to be incorporated when using WUA metrics to describe fish habitat.

While many studies recognize the importance of accurate topography measurements to create a representative 2D hydraulic model (e.g. Waddle et al. 2010), our results suggest that model results can change dramatically (in this case up to 39% change in velocity averaged over an 18-m length of stream) due to differences in topography measurement and interpolation techniques. For this study, the topography model using the total station data was used because this is the model that was previously calibrated and because the topographic models created using the TLS data under represented the below water surface topography and resulted in shallower flow depths. It should be noted that the points surveyed using the total station consistently underestimated individual rock volumes compared to TLS (average 55% smaller; Resop 2010). This over simplification of boulder shape has implications for model results, although an accurate representation of below water surface topography was deemed more important in this study.

Landscape ecology metrics appeared to describe the distribution of low velocity outlier areas in this study, but there was no relationship between these metrics and brook trout population data. It is possible that these metrics could prove useful in describing habitat connectivity, but would likely be more useful if the model and clustering scheme better represented the influence of boulders on in-stream habitat.

3.6. Acknowledgements

3.7. References

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Chapter 4:   Combined Influence of Bank Vegetation and Watershed Imperviousness on Stream Temperature

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4.1. Abstract

We investigated the combined influence of urbanization and riparian vegetation on temperature in 27 small streams (0.1 km^2 – 122 km^2) in paired forested/nonforested stream reaches in southeastern Pennsylvania, Maryland, and Delaware. The watersheds contributing to the paired reaches spanned an urban land use gradient (total imperviousness ranging from 1% to 66%). Riparian vegetation and watershed-scale urbanization both influence stream temperature which has profound impacts on in-stream ecosystems. Generally, increased urbanization results in higher stream temperature as does a reduction in riparian forests. We measured temperature ($\pm$ 0.2 $^\text{o}$C) hourly at the downstream end of each paired reach for at least one year. The paired reaches are nearly contiguous, so all significant watershed-scale variables are held constant, allowing for comparison of stream temperature due to reach-scale changes in riparian vegetation and channel morphology. In general, forested reaches had lower maximum daily temperature, while the diurnal temperature range was significantly greater in nonforested reaches in the summer and significantly greater in forested reaches in the winter. The increase in stream temperature with shade reduction suggests the importance of riparian buffer preservation to maintain shading and channel form. The influence of bank vegetation on stream temperature decreased with urban landuse and watershed area.

Keywords: stream temperature, urbanization, riparian vegetation

4.2. Introduction

Stream temperature is an important factor that directly and indirectly affects almost all aspects of stream ecology (Hynes 1970). Increased stream temperatures can shift biological communities away from more tolerant species, thereby adversely affecting biological diversity (Krause et al. 2004). Elevated water temperature can also indirectly harm fish and macroinvertebrates by lowering dissolved oxygen (DO) concentrations.

Stream temperature is proportional to heat energy added to the stream (heat load) and inversely proportional to discharge (Poole and Berman 2001); any activities that alter the timing or magnitude of heat load or flow regime can be expected to alter stream temperature. Different heat exchange processes in stream environments occur at the air/water interface and streambed/water interface (Caissie 2006). At the air/water interface, heat flux is primarily driven by solar radiation (net short-wave radiation), net long-wave radiation, evaporative heat flux, and convective heat transfer. Heat flux at the streambed/water interface is a function of conduction and of advective heat transfer through groundwater or hyporheic exchange. Other sources of energy exchange include friction and precipitation; however, the contribution of these sources to the overall stream heat energy exchange is generally small compared to the other components.

Riparian vegetation characteristics such as height, density, and proximity to the channel affect the amount of solar radiation that reaches the stream, but also can affect near-stream wind speed, which in turn decreases atmospheric heat exchange (Poole and Berman 2001). The influence of bank vegetation on stream temperature has been well documented (Yoshida et al. 2005; Webb and Crisp 2006; Hannah et al. 2008; Davies-Colley et al. 2009). Stream restoration activities such as the planting of riparian buffers often occur at a reach scale (100-200 m). At this scale, shading is expected to influence maximum daily temperature, but not overall mean and minimum daily temperature (Johnson 2004).

Anthropogenic stressors associated with urban land use, such as increased watershed imperviousness, destruction of riparian vegetation, and increased siltation, are expected to alter stream temperature regimes (Nelson and Palmer 2007). The increase in impervious surfaces associated with urbanization affects both stream discharge and heat load. Increased imperviousness leads to a number of hydrologic changes including reduced infiltration, which in

turn can reduce baseflow and increase the flashiness of urban streams. The runoff over heated parking lots and other imperious surfaces is a source of heat load to urban streams (Van Buren et al. 2000; Herb et al. 2008). Generally, increased urbanization has been found to result in higher stream temperatures (LeBlanc et al. 1997; Poole and Berman 2001; Krause et al. 2004). Elevated stream temperature in urban streams is commonly attributed to changes to riparian vegetation, stream width, streamflow, and groundwater (Krause et al. 2004). Temperature surges caused by thermal enhancement of stormwater runoff over impervious surfaces are also an important mechanism for thermal pollution in urban areas (Van Buren et al. 2000; Nelson and Palmer 2007). Other potential factors include the removal or introduction of wetlands or ponds, and point source thermal discharges (LeBlanc et al. 1997).

Here we present analyses of stream temperature in paired stream reaches with and without riparian forests, with watersheds ranging from rural to highly urban in Pennsylvania, Maryland, and Delaware. Our study was designed to answer two questions concerning stream temperature: 1) Does riparian vegetation influence stream temperature characteristics at a reach scale?; and, 2) If riparian vegetation does influence reach-scale stream temperature, does this influence change depending on level of watershed urbanization?

4.3. Study Sites

We used data from 27 paired forested/nonforested stream reaches approximately 100-250 m in length located in southeastern Pennsylvania, northern Maryland, and Delaware (Figure 4-1). Each stream reach pair was nearly contiguous with the forested reach upstream of the nonforested reach in 18 of the 27 pairs (Table 4-1). Sites were chosen based on two criteria: (1) no major tributaries could enter between reaches; and, (2) no-major disturbances such as livestock access or forest harvesting (Hession et al. 2003; Sweeney et al. 2004). While the paired reaches in the previous geomorphic study did not abut each other, in this temperature study, two of the study reaches were adjacent with one temperature probe serving as both the downstream probe for the upper reach and the upstream probe for the lower reach.

The paired-reach design using nearly contiguous study reaches allows us to assume watershed characteristics are equivalent between forested and nonforested reaches in each pair (Hession et al. 2003). The watersheds used in this study covered a gradient of urbanization (5%

to 66% impervious cover) from fragmented mixed-hardwood deciduous forest in rural areas to suburban and highly urbanized land cover in and around Philadelphia, PA. Each watershed was classified as urban or nonurban based on the percent impervious cover calculated by Hession et al. (2003). The natural break in watershed percent imperviousness that occurred between approximately 12% and 30% served as the cutoff between the two classes. Watershed areas ranged from 0.13-122 km^2 and were located in the Piedmont Uplands formed primarily on schist and the Piedmont lowlands formed on sandstone and shale. Precipitation in this area is evenly distributed throughout the year (annual average 1170 mm).
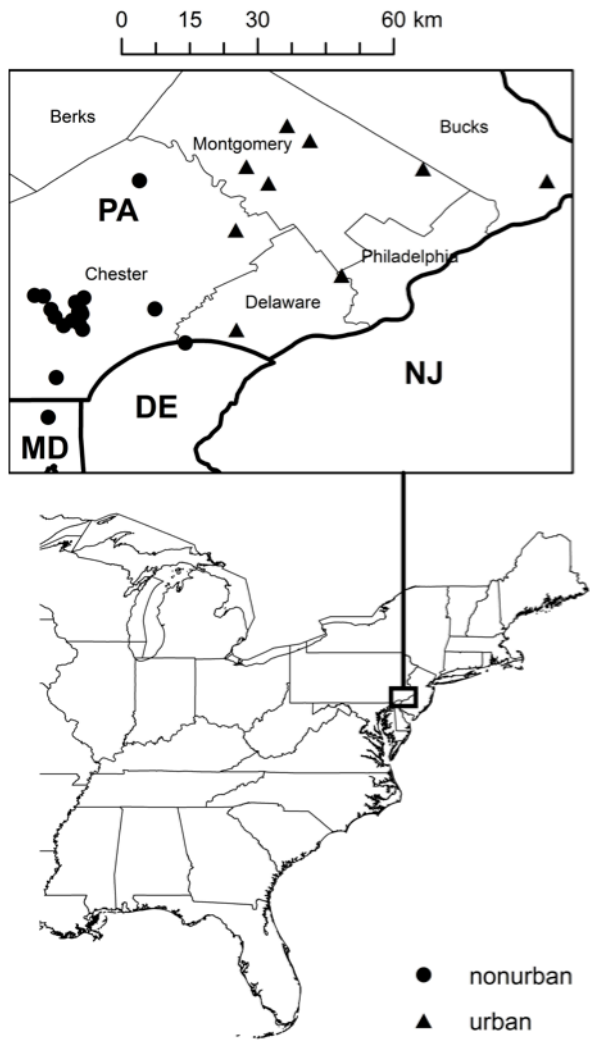
Figure 4-1. Location of paired stream temperature monitoring sites in southwestern Pennsylvania (PA), Maryland (MD), and Delaware (DE).

Table 4-1. Study site characteristics and sampling periods.

Site Name	Drainage Area (km²)	Reach Length (m)	Width (m)	Depth (m)	Cross-sectional Area (m²)	Percent Canopy	Start	End
			Mean Bankfull Channel Dimensions[a]					
Nonurban								
Jocks Run*	0.1	102/201[c]	4.5/2.0	0.14/0.07	0.7/0.1	nm[d]	3/17/97	12/31/98
Hannum's Run*	0.7	203/148	3.9/1.9	0.18/0.22	0.8/0.4	nm	12/16/98	11/4/99
Brandywine Tributary*	0.8	99/145	3.8/1.2	0.27/0.32	1.0/0.4	nm	5/27/97	4/1/98
West Branch White Clay*	1.4	150/151	3.7/2.1	0.25/0.31	0.9/0.7	nm	6/26/98	7/22/99
West's Run*	1.5	170/153	5.7/2.5	0.26/0.27	1.5/0.6	nm	3/25/97	7/2/98
Moorehead's Run*	1.9	147/153	5.2/2.6	0.34/0.3	1.8/0.8	78/10[e]	3/28/97	7/7/98
Big Springs*	2.2	150/147[b]	5.5/4	0.29/0.33	1.7/1.3	50/50[e]	12/3/97	12/31/98
Teeter's Run*	2.5	244/277	3.8/2.5	0.33/0.28	1.2/0.7	nm	4/14/97	7/16/98
Grammie's Run*	3.3	208/208	9/4.8	0.74/0.52	6.5/2.5	nm	10/9/98	12/31/99
Sharitz Creek	4.4	147/191	6.8/3.8	0.31/0.5	2.0/1.9	nm	10/9/98	12/31/99
White Clay Creek*	7.1	155/205	6.9/6.4	0.45/0.65	3.0/3.9	80/21[f]	1/1/98	12/31/98
Birch Run	7.6	204/200	9.1/4.5	0.37/0.27	3.4/1.2	90/28[f]	6/18/98	9/29/99
Beaver Run*	12.3	147/197	7.4/6	0.32/0.33	2.4/2.0	94/0[g]	9/25/98	11/2/00
Fisher's Run*	13.3	146/140	8.3/5.5	0.41/0.55	3.5/3.0	83/63[f]	6/9/98	8/3/99
Pocopson Creek	21.3	150/150	10.1/7.3	0.55/0.51	5.5/3.9	87/60[f]	7/10/98	7/19/99
Doe Run	25.3	264/264	11/8.8	0.52/0.38	5.5/3.4	86/44[f]	10/9/98	12/31/99
Doe Laurels Creek*	50.9	150/200	15.4/12.4	0.71/0.72	11/9.1	nm	2/24/99	3/5/00
Buck and Doe	122.2	189/200	24.9/16.4	0.78/0.88	18/14	80/45[f]	9/11/98	1/13/00
Urban								
Cobbs Creek*	0.4	150/200[c]	15.4/12.4	0.71/0.72	2.0/0.9	65/0[f]	6/5/99	4/5/01
Eagleville Run	1.0	132/81	5.2/3.3	0.38/0.28	2.0/1.8	70/0[f]	10/1/98	11/23/99
Donny Brook	3.2	122/131	5.4/4.8	0.38/0.38	2.3/0.8	97/0[f]	10/17/98	11/22/99
W. B. Skippack Creek	3.7	103/95	6.1/2.8	0.37/0.27	2.6/2.0	nm	10/8/98	11/22/99
Pennypack Creek*	4.6	209/112[b]	7.8/5.1	0.34/0.36	2.8/2.0	10/0[f]	12/2/99	10/26/00
Towamencin Creek	6.0	79/105[b]	7.1/5.5	0.4/0.37	2.9/1.6	78/21[f]	12/15/99	10/26/00
Green Creek*	8.4	133/100[b]	10.9/4.5	0.36/0.39	4.5/2.6	90/0[f]	11/12/99	9/30/00
Little Valley Creek*	13.1	126/105[c]	8.4/5.3	0.53/0.49	1.9/1.9	97/9[f]	12/7/99	4/5/01
Queen Anne Creek*	15.6	95/146	6.2/3.9	0.3/0.47	0.46/0.34	94/46[f]	10/6/98	9/2/00

Note: Slash indicates forested/nonforested parameter values. *Forested reach is upstream of nonforested reach; [a] Average of five or six bankfull values determined at each site; [b] no upstream temperature probe; [c] downstream forested reach temperature probe was used as upstream nonforested probe; [d] nm = not measured; [e] canopy measured by visual estimate; [f] canopy measured using mean of 6-10 images; [g] canopy measured by averaging 3-6 transects per reach

4.4. Methods

4.4.1. Stream Morphology

Stream reach geomorphic data were collected as part of two previous studies during base flow periods in the summers of 1997 through 1999 (Hession et al. 2003; Sweeney et al. 2004). Study reaches were 10 to 20 bankfull widths in length and all survey measurements were made using a laser level and survey tapes. Detailed geomorphic survey methods are outlined in Cianfrani et al. (2006), Hession et al. (2003), and Pizzuto et al. (2000).

4.4.2. Stream and Air Temperature

Stream temperature was measured hourly or quarter-hourly at the upstream and downstream cross section of each reach for approximately one year between 1997-2001 using Optic StowAway temperature data loggers (Onset Computer Corporation, Bourne, MA) with an accuracy $\pm 0.2\ ^{\circ}C$ (Table 4-1). This resulted in a total of four temperature probes per reach with the exception of Cobbs Creek and Little Valley Creek where the forested downstream probe was used as the nonforested upstream probe. There were no upstream probes for Green Creek, Towamencin Creek, or Pennypack Creek. Quarter-hourly (15 min) temperature data were adjusted to an hourly time step. Hourly air temperature was acquired from the U.S. Department of Commerce National Climatic Data Center (NCDC) Willow Grove NAS (WBAN 14793) station for the entire study period (1997-2001). Figure 4-2 shows the monthly average maximum and minimum daily air temperature and total monthly precipitation over the study period.
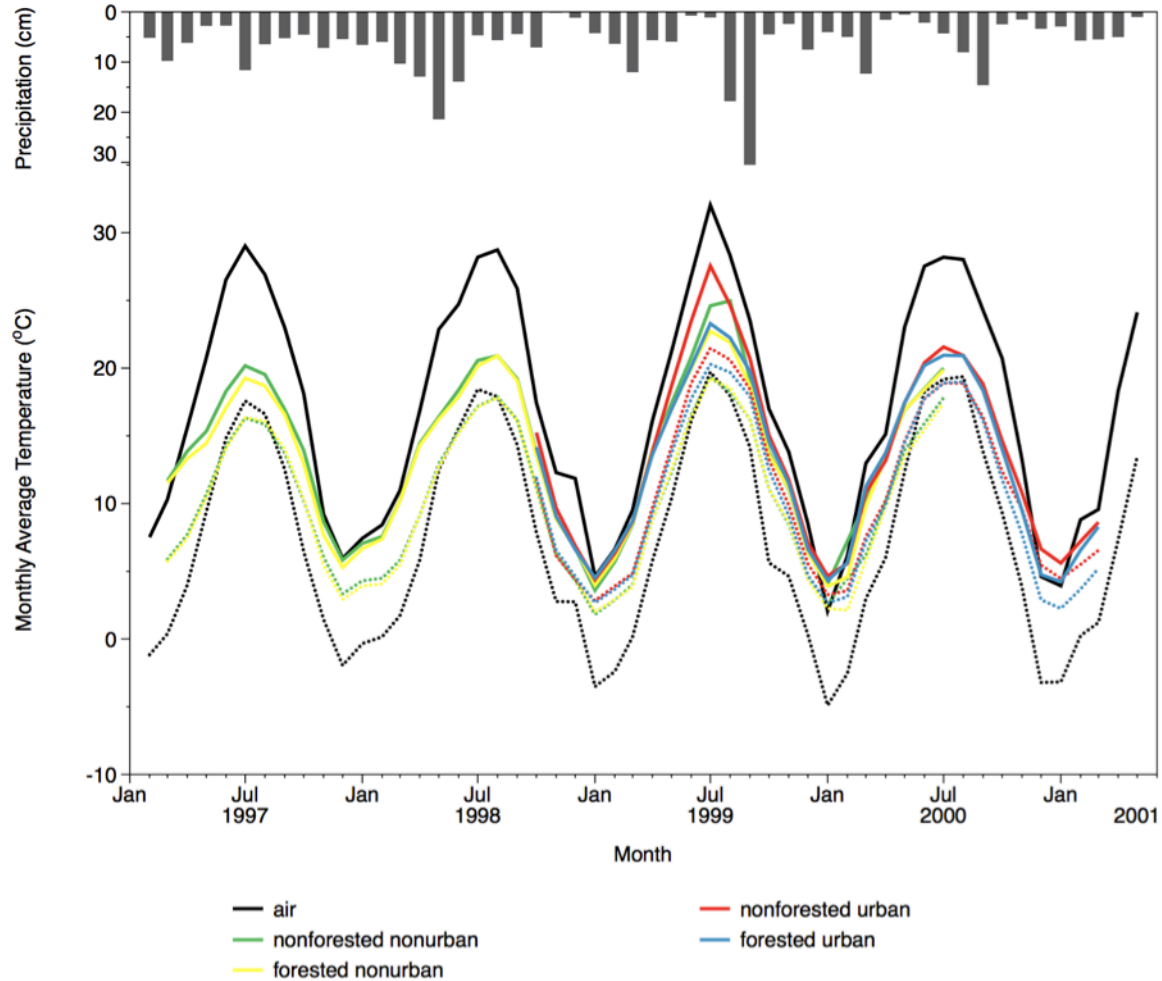
Figure 4-2. Monthly average of daily maximum and minimum air and water temperature over the study period. Dashed lines indicate monthly means of daily minimum temperatures, and solid lines indicate monthly means of daily maximum temperatures.

4.4.3. Statistical Analysis

Differences in temperature characteristics between forested and nonforested reaches were examined using a linear-mixed model with repeated measures (time series) and random effects (stream) using the MIXED procedure in SAS (SAS Institute 2009). The model was used to test for significant differences between forested and nonforested temperature characteristics after accounting for seasonal differences. Seasons were defined by solstice and equinox dates: winter (December – March), spring (March – June), summer (June – September), fall (September – December). Linear mixed modeling has been shown to improve temperature prediction accuracy

by accounting for nonlinear relationships between the response (temperature characteristic) and predictor variables (i.e. urbanization and riparian vegetation characteristics), and by accounting for spatial autocorrelation among sampling sites (Wehrly et al. 2009). Applied to time-series data, a linear-mixed model can account for the autocorrelation of subsequent measurements (Kuzyakova et al. 2006).

Using the downstream probes only for each reach, daily stream temperature statistics were calculated including maximum and minimum temperature (T_{max} and T_{min}, respectively), temperature range (T_{range}), and mean temperature (T_{mean}). The rate of temperature change per unit length (T_{rate}) was calculated as the difference between the upstream and downstream temperature probe divided by the reach length. The effect of urban land use was included in this analysis by testing for significant differences between urban and nonurban watersheds with respect to the differences in daily stream temperature characteristics and after accounting for seasonal differences.

Before testing for effects of riparian vegetation, the effect of seasonality and urban classification on the differences in forested minus nonforested daily temperature characteristics was tested. All statistical tests used an $\alpha = 0.05$ for significance. There were significant seasonal effects for all temperature characteristics (T_{max}, T_{mean}, T_{min}, T_{range}, and T_{rate}). The interaction of season and urban classification was also significant across all temperature characteristics. Subsequently, data were divided into urban and nonurban datasets and analyzed separately, taking into account the effect of seasonality of daily temperature characteristics. A similar analysis was conducted to test for the significance of the location of the forested reach and there was no significant difference between sites where the forested reach was upstream and sites where the nonforested reach was upstream with respect to the forested minus nonforested difference in temperature characteristics (T_{max}, T_{mean}, T_{min}, T_{range}, and T_{rate}).

Temperature extremes were analyzed by calculating temperature duration curves for T_{max} at the downstream probe for urban and nonurban, and forested and nonforested sites. Relative durations at extreme summer temperatures were calculated for each scenario and qualitatively compared.

Finally, the maximum daily temperature (T_{max}) over the whole study period was fit to a sine wave:

$$T_{\max}(t) = A\sin(\omega t + \phi) + D \quad (1)$$

where A is the amplitude, ω is the frequency, ϕ is the phase, and D is the offset from 0. Because daily maximum temperature followed an annual cycle, ω and ϕ were assumed to be constant for all models (urban forested, nonurban forested, urban nonforested, and nonurban nonforested). Therefore time was linearized by:

$$DAY_{trans} = \sin(2\pi \times \frac{DAY}{364.6406} - 0.6688) \quad (2)$$

where DAY is the number of days from the study start. Substituting equation (2) into equation (1) results in the linear equation:

$$T_{\max}(DAY_{trans}) = A \times DAY_{trans} + D \quad (3)$$

The model parameters *A* and *D* were then calculated using a linear mixed model with repeated measures and random effects similar to the model used to test for seasonal effects. This model provides insight into annual T_{max} cycle under each scenario.

4.5. Results

4.5.1. Seasonal Temperature Characteristics

In general, across all seasons, mean daily nonforested temperature characteristics (T_{mean}, T_{max}, and T_{min}) were warmer than forested temperature characteristics. Differences between forested and nonforested T_{mean} were small, but significant in both urban and nonurban watersheds (Figure 4-3; Table 4-2). Similarly, nonforested T_{max} was warmer than the paired forested reach T_{max} although differences were small in winter in urban watersheds and not significant in winter in nonurban watersheds. Seasonal mean T_{min} was warmer for all seasons in urban nonforested sites, and for winter and spring in nonurban nonforested sites (Figure 4-4; Table 4-2). Forested and nonforested diel temperature variations (T_{range}) were most significantly different in winter when forested ranges were greater than nonforested and summer when nonforested ranges were

greater than forested. In winter, forested reaches cooled slower than nonforested reaches in urban watersheds and warmed in nonurban watersheds. In spring and summer in urban watersheds, water warmed throughout nonforested reaches and cooled slightly in forested sites. For all other seasons, T_{rate} (^{o}C/km) was more negative in nonforested reaches than forested in both urban and nonurban watersheds (Table 4-2).

Table 4-2. Seasonal means of daily temperature characteristics.

	Urban				Nonurban			
Daily	Winter	Spring	Summer	Autumn	Winter	Spring	Summer	Autumn
T_{max} (^{o}C)	5.91/6.11	15.7/16.14	21.3/22.8	11.69/12.3	6.06/6.12[a]	15.3/15.61	20.32/21.28	10.75/11.07
T_{min} (^{o}C)	3.42/4.04	12.37/12.74	19.02/19.43	9.5/10.07	3.1/3.34	10.92/11.09	17.28/17.29[a]	8.28/8.25[a]
T_{range} (^{o}C)	2.5/2.06	3.33/3.40[a]	2.29/3.37	2.19/2.23[a]	2.96/2.78	4.38/4.51	3.04/4	2.48/2.82
T_{mean} (^{o}C)	4.59/5.02	13.94/14.33	20.12/20.92	10.57/11.12	4.48/4.65	12.93/13.15	18.72/19.03	9.48/9.58
T_{rate} (^{o}C/km)	-1.2/-3.5	-0.94/3.6	-0.09/2.9	-0.19/-3.0	1.9/-0.33	0.33/-0.86	-2.4/-4.5	2.1/-1.2

Note: Slash indicates forested/nonforested parameter values.
[a] Riparian vegetation was not significant (α=0.05)
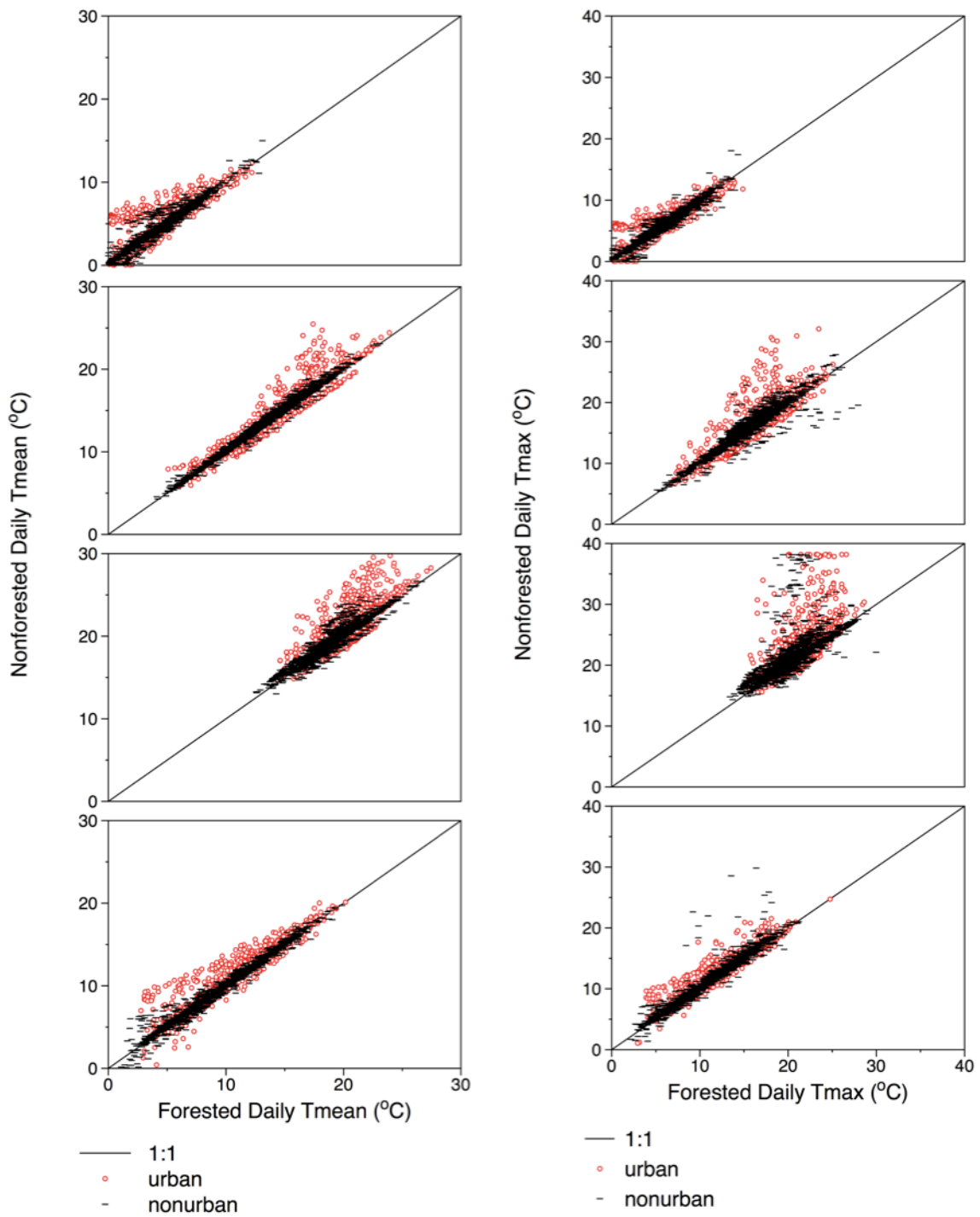
Figure 4-3. Daily temperature statistics (T_{mean} and T_{max}) at downstream probe in forested vs. nonforested riparian reaches by seasons (from top: winter, spring, summer and autumn).
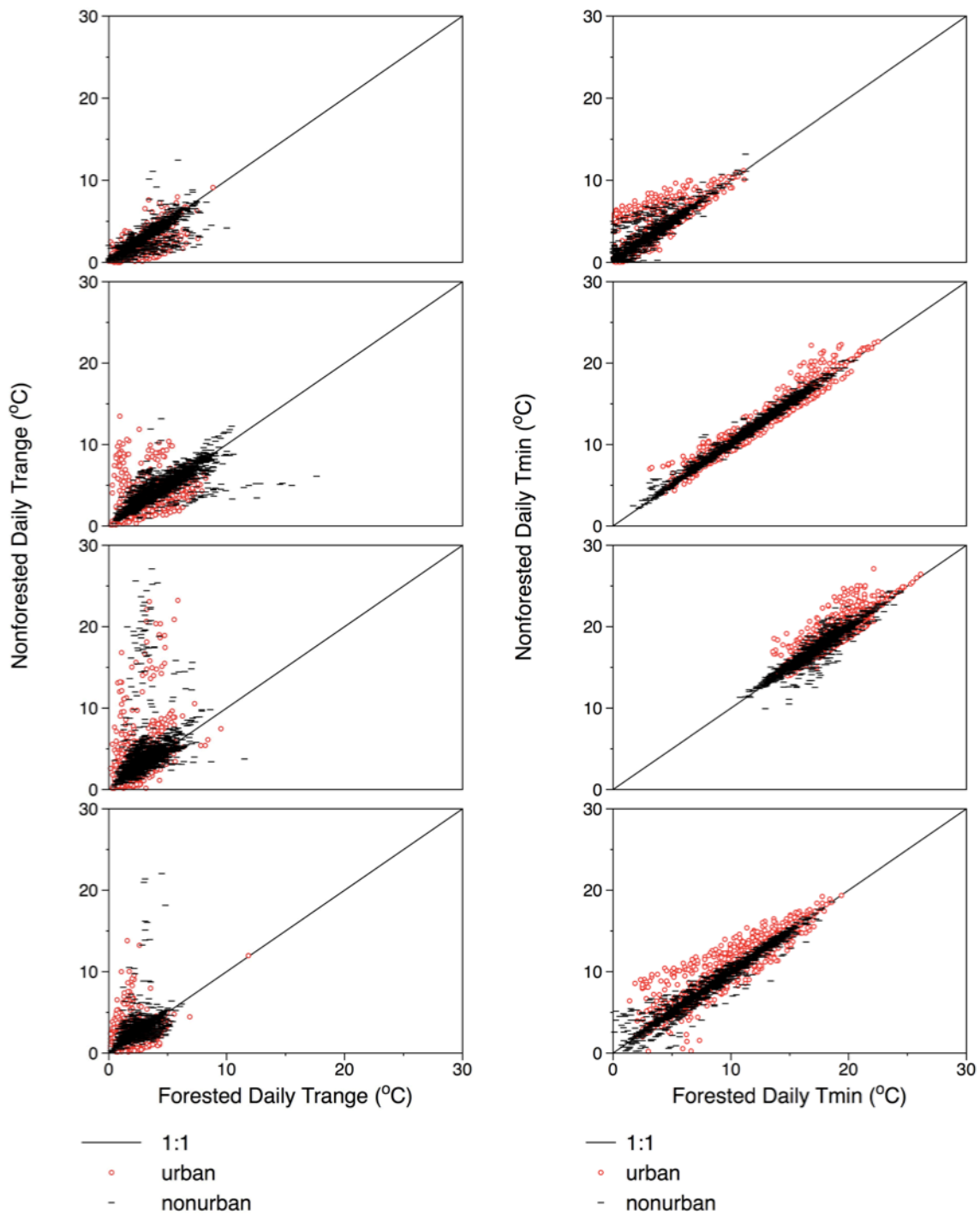
Figure 4-4. Daily temperature characteristics (T_{min} and T_{range}) at downstream probe in forested vs. nonforested riparian reaches by seasons (from top: winter, spring, summer and autumn).

4.5.2. Temperature Extremes

Although daily temperature characteristics were generally warmer in nonforested reaches than forested reaches, mean differences were typically within 1-2 ^{o}C. However, at individual sites, extreme high temperatures occurring during the summer months are of particular concern for aquatic biota (i.e. Krause et al. 2004). Study period maximum daily temperature cumulative frequency distributions were plotted to illustrate the impact of riparian vegetation and watershed urbanization on extreme maximum summer temperatures (Figure 4-5; Table 4-3). Maximum daily temperatures in nonforested sites exceeded the Pennsylvania state water temperature standard (30.6 ^{o}C) for 60 days in three streams in urban watersheds (38 days in June-August 1999 in W.B. Skippack Creek; 20 days in July-August 1999 in Donny Brook; and two days in June 1999 in Cobb's Creek). In nonurban watersheds, maximum daily temperatures in nonforested sites exceeded the state standard for 47 days in three streams in July and August 1999 (39 days in Brandywine Tributary, 4 days in Doe Run, and 4 days in Jock's Run). While extreme summer temperatures occurred in both urban and nonurban watersheds in nonforested sites, in forested sites, maximum daily temperatures did not exceed the state temperature standard. Using the T_{max} 90th percentile for all study sites (22°C) as the cutoff for defining extreme maximum daily temperature, over the study period, both the mean number of days and the mean duration of elevated temperatures were greater in urban sites than nonurban sites and greater in nonforested sites than forested sites (Table 4-4).

Table 4-3. Cumulative frequency distribution of maximum daily temperature (^{o}C).

% Equaled or exceeded	Urban	Nonurban	All Sites
5%	23.53/25.76	22.51/23.68	23.70
10%	22.16/23.67	21.21/21.88	21.98
25%	19.22/19.63	18.76/19.23	19.17
50%	13.59/13.68	14.26/14.56	14.13
75%	7.98/8.35	8.39/8.49	8.38
90%	4.69/5.26	5.36/5.61	5.28
95%	2.84/3.12	3.89/4.06	3.56

Note: Slash indicates forested/nonforested parameter values.
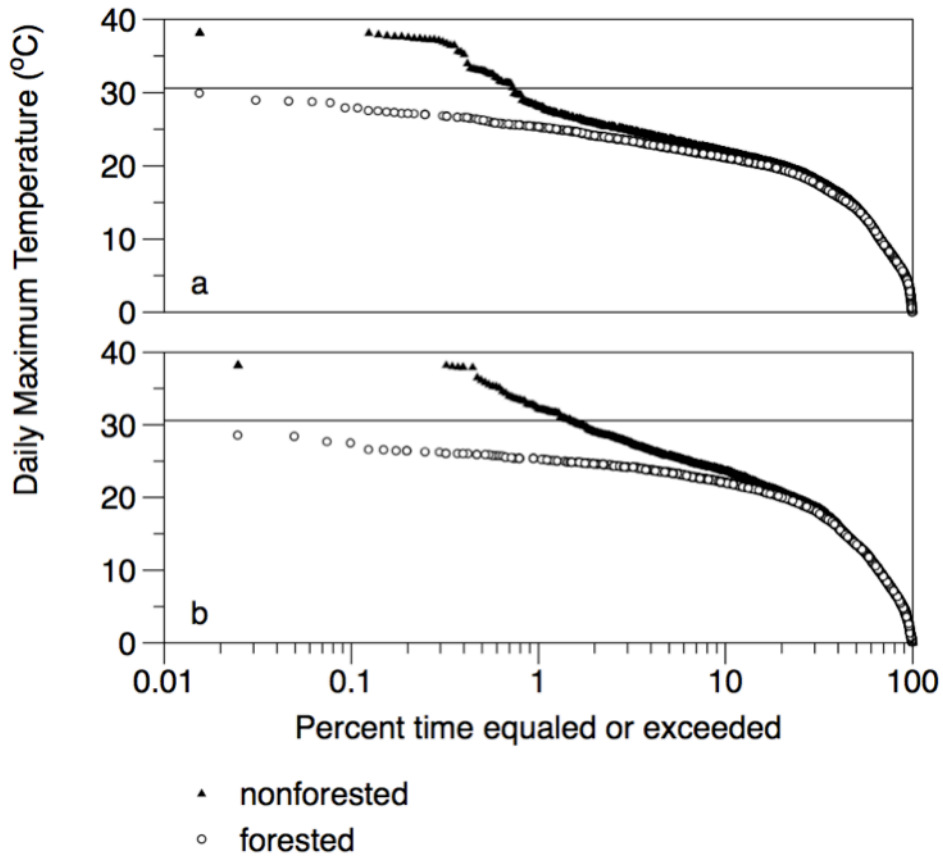
Figure 4-5. Percent time equaled or exceeded of maximum daily temperature for a) urban sites and b) nonurban sites. 30.6 °C is the maximum Pennsylvania water temperature standard.

Table 4-4. Mean number of days and hours per day stream temperature exceeded 22°C (90th percentile of maximum daily temperature distribution for all sites) and the Pennsylvania state temperature standard (30.6°C).

		Urban	Nonurban
Duration above 22 °C	sites	9/9	18/18
	days	48/66	31/41
	mean hrs/day	10.2/11.6	7.2/7.5
Duration above 30.6 °C	sites	-/3	-/3
	days	-/30	-/16
	mean hrs/day	-/3.3	-/1.8

Note: Slash indicates forested/nonforested parameter values.

4.5.3. Temperature Time Series

        To evaluate the impact of riparian vegetation and watershed urbanization on annual temperature patterns, the linearized form of maximum daily temperature cycle was fit with four sine equations following seasonal variations for forested urban, nonforested urban, forested nonurban and nonforested nonurban (Figure 4-6). The effects of urbanization and riparian forest classification were significant; however, although the interaction between urban and riparian classification slightly improved the model fit, the interaction was not statistically significant. When watershed area was added, model fit improved; the modified sine equation better fit the spread seen in summer maximum daily temperatures when watershed size was incorporated. Urban watersheds with nonforested riparian areas had the greatest offset and amplitude and nonurban watersheds with forested riparian areas had the smallest offset and amplitude (Table 4-5). Although many of the extreme T_{max} measurements occurred in smaller watersheds, in general, increased watershed area resulted in increased offset and amplitude. Forested riparian reaches had a greater increase in both amplitude and offset with increased watershed areas.
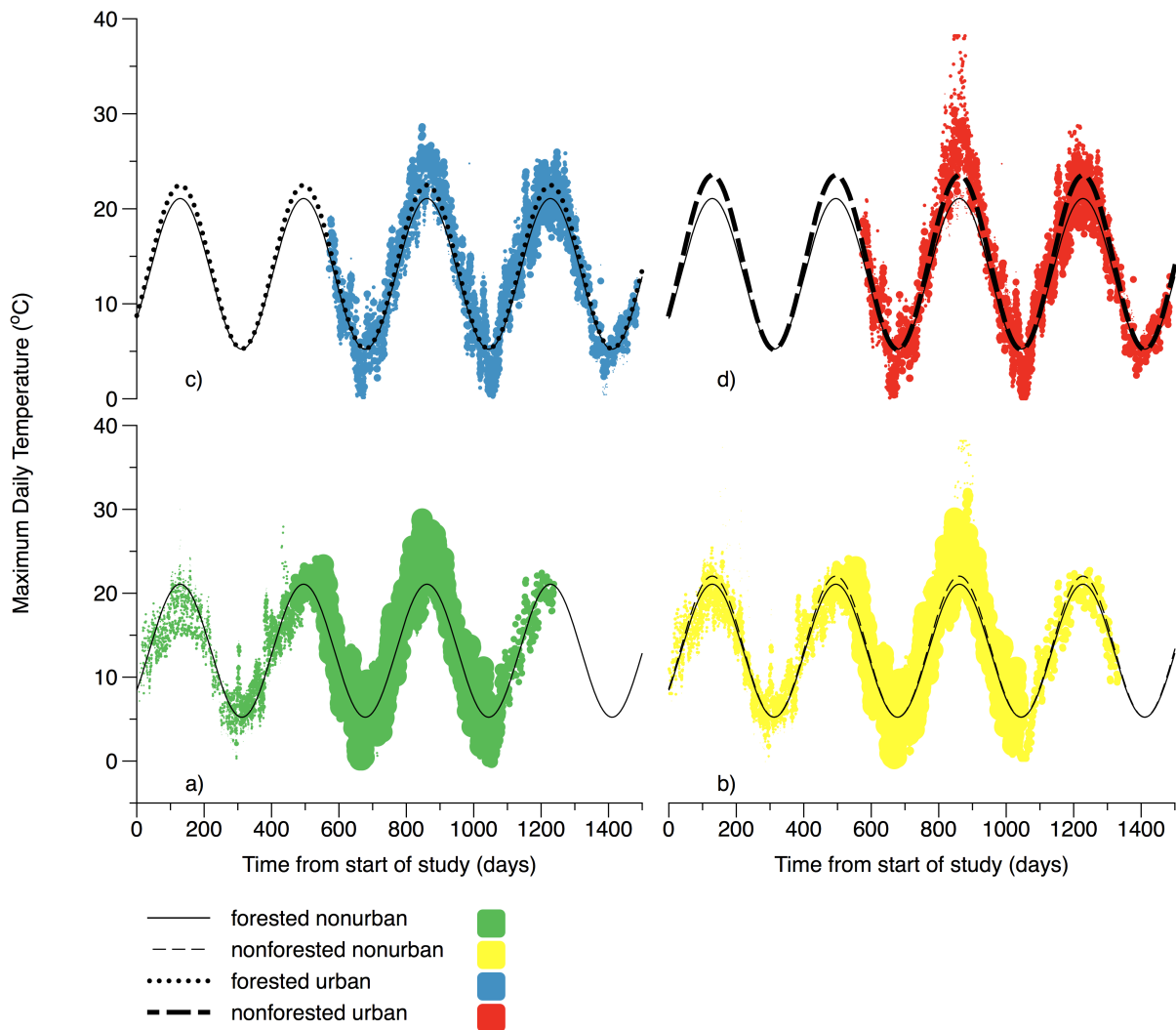
Figure 4-6. Maximum daily temperature over the study period for all sites with fitted sine wave models incorporating urban/riparian vegetation characteristics: a) forested nonurban, b) nonforested nonurban, c) forested urban, and d) nonforested urban. Points are scaled by watershed area.

Table 4-5. Maximum daily temperature time series model parameters for urban/riparian vegetation model without and including watershed area.

	Urban/Riparian Vegetation		Including Watershed Area	
	Offset	**Amplitude**	**Offset**	**Amplitude**
Urban/Forested	13.876	8.691	13.681 + 0.0137(WS_area)	8.374 + 0.0292(WS_area)
Urban/Nonforested	14.362	9.165	14.430 + 0.00761(WS_area)	9.147 + 0.0211(WS_area)
Nonurban/Forested	13.152	7.929	13.017 + 0.0137(WS_area)	7.536 + 0.0292(WS_area)
Nonurban/Nonforested	13.648	8.402	13.481 + 0.00761(WS_area)	8.139 + 0.0211(WS_area)

4.6. Discussion

In general, stream temperature was warmest in urban watersheds without riparian forest cover, especially during summer. Over the study period, the annual time series of daily T_{max} in urban streams was shifted up by 0.7 $^{\circ}$C and had a larger amplitude,1 $^{\circ}$C and 0.7 $^{\circ}$C in forested and nonforested sites, respectively than nonurban streams. Winter T_{max} values were similar for regardless of riparian vegetation and watershed urbanization; however, summer T_{max} values were up to 1.8 $^{\circ}$C warmer in nonforested urban reaches than in forested nonurban reaches. The difference in seasonal mean T_{max} between forested and nonforested reaches was most dramatic in summer (1.5 $^{\circ}$C in urban watersheds and 1 $^{\circ}$C in nonurban watersheds).

There was no statistically significant seasonal effect on timing of maximum daily temperature between either forested and nonforested sites or air temperature and water temperature indicating that either the hourly time step was too large to capture any lag or that the relatively small stream size allowed the water temperature to react quickly to climatic conditions. Over all of the study sites, the differences in T_{max} timing between forested and nonforested reaches on either a daily or annual cycle were not significant; water temperature was in phase with air temperature and the difference between time of maximum air temperature and time of maximum water temperature was not significant across reaches. This study did not incorporate the effect of large-scale topographic influences on stream shading such as stream aspect or valley width. Because of the paired-reach design, these characteristics were assumed to be similar among nearly contiguous reaches. The effect of width to depth ratio was incorporated in the watershed characterization of urban or nonurban and riparian vegetation type. Previous work suggested that the paired forested reaches were wider than nonforested reaches used in this study (Hession et al. 2003; Sweeney et al. 2004).

Watershed size had a significant effect on the maximum daily temperature baseline and amplitude indicating that the influence of riparian vegetation decreases as watershed size increases. This result is likely due both to the changes in bankfull width with increasing watershed size (Hession et al. 2003). As stream width increases, shading from riparian vegetation covers less of the total surface area allowing more solar radiation to reach the water surface.

Because stream temperature is a function of heat load and discharge (i.e. Poole and Berman, 2001), increased heat load (reduced shading) and increased discharge as watershed size increases, reduces the cooling and buffering influence of riparian vegetation. Although the study sites did not have continuous discharge measurements over the study period, very low summer flows were observed in at least four of the study sites that corresponded with the study sites that had excessive summer stream temperatures ($> 30.6\,^{\circ}\mathrm{C}$). These sites also corresponded with the lowest reported linear fish densities likely due to a combination of poor water quality, high temperatures, and very little available habitat (Horwitz et al. 2008).

Bank vegetation had both a cooling and a buffering effect on steam temperatures. When comparing temperature cycles, a difference in means denotes cooling, while a difference in range denotes buffering (Arrigoni et al. 2008). Summer diel temperature ranges were larger in nonforested reaches than forested reaches ($1.1\,^{\circ}\mathrm{C}$ and $1\,^{\circ}\mathrm{C}$ in urban and nonurban sites, respectively) indicating a buffering influence of riparian forest vegetation, while seasonal means of daily mean temperature were consistently warmer in nonforested sites indicating a cooling influence in forested sites. It is difficult to tease out the exact processes driving these responses, although shading is considered to be an insulating (cooling) process and groundwater or hyporheic inflow/exchange is considered a buffering process (Poole and Berman 2001).

4.7. Conclusions

Both riparian vegetation and watershed characteristics had significant impacts on stream temperature. In general, stream temperature was warmer in nonforested reaches than forested reaches and was warmer in urban watersheds than nonurban watersheds. For forested reaches in urban watersheds, the baseline stream temperature was elevated compared to nonurban watersheds, but was cooled and buffered when compared to the paired nonforested urban stream temperatures illustrating the importance of riparian vegetation in mitigating stream temperature increases from land use change. This impact was significant at the reach scale (100-200 m) used in this study. Seasonal means were independent of which reach (forested or nonforested) was upstream indicating that the influence of riparian vegetation on stream temperature in these mixed land use streams was local. The overall impact of riparian vegetation on downstream temperatures is unknown. The effect of shading in this study was similar to the results presented

by Johnson (2004) who found that shading short sections of stream had a larger effect on maximum daily temperature than minimum or mean daily temperature by decreasing instantaneous midday energy fluxes.

Riparian vegetation was also important in decreasing extreme summer temperature frequency and duration. Both maximum daily temperatures and the frequency and duration of elevated temperatures were reduced in forested reaches implying that at this scale, riparian vegetation is important for maintaining temperature regimes suitable for thermally sensitive organisms. Previous work by Horwitz et al. (2008) on these study reaches found local, reach-level effects of riparian vegetation on the abundance of several fish species, although many fish species showed similar linear densities in forested and nonforested reaches. While recent research on stream temperature in urban watersheds (e.g. Nelson and Palmer 2007) has emphasized the effect of runoff over hot pavement, the effect of temperature surges as a result short duration summer storms could not be separated in this study because of the lack of corresponding local precipitation data for all sites. The weather station used for air temperature was up to 40 km away from study sites; therefore, local summer storms could not correlated to temperature surges. The approach used in this study can be applied to studies evaluating the effect on local stream temperature due to changes in urbanization or riparian vegetation scenarios.

4.8. Acknowledgments

4.9. References

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Chapter 5: Conclusions and Future Research

This research focuses on the interdisciplinary linkage between aquatic ecology and engineering necessary for successful stream habitat restoration. While many restoration activities can directly or indirectly influence aquatic habitat, this research focuses on the influence of channel morphology, level of watershed urbanization and riparian vegetation on local (meso- and macro-) in-stream habitat characteristics. Moving beyond one-dimensional (1D) averaged representations of fish habitat, this research aimed to explore the distribution of two-dimensional (2D) flow complexity metrics at the mesohabitat scale as explanatory variables for fish habitat preferences and as potential metrics to evaluate habitat restoration design. The 2D distribution of flow characteristics was then investigated further to quantify areas of flow refugia and the relationship between these areas, traditional measures of habitat quality, and the fish biomass. Flow complexity in the vicinity of flow obstructions (in this case, boulders) was investigated further using patch classification and landscape ecology metrics. Finally, the relative influence of riparian vegetation on stream temperature (another important habitat characteristic) in urban and nonurban watersheds was investigated. Overall, changes to channel morphology and riparian vegetation had a local effect on stream habitat (temperature and hydraulic habitat) that was demonstrated by temperature fluctuations and brook trout mass.

5.1. 2D Hydraulic Modeling for Habitat Analysis

While widely used to evaluate physical habitat, 2D models are limited in their ability to represent complex flows. A model is a representation of reality, and any hydraulic model is a generalization of a complex river or stream. There are two levels of approximation that tend to compound each other. The first is an abstraction of physical reality to a mathematical formulation, and the second is the transformation of mathematical statements to computer arithmetic (Steffler and Blackburn, 2002). The influence of these approximations is dependent on the system of interest and the scale of the system's characteristics that are being modeled.

For example, studies of near boundary flow conditions would require a model capable of modeling the flow characteristics at this scale. As the hydraulic characteristics to be modeled

become more complex, the modeling framework also becomes more complex. In general, the detail of data collection increases as the model's ability to handle complex flows increases.

Qualitatively, even at moderate flows, the boulders within the Staunton River are mostly exposed. There are few large rocks in the deep pools, and wakes from submerged rocks may not play a large role in the overall hydraulic complexity. Wakes behind exposed boulders are expected to play the largest role in flow complexity at low to moderate flows in this system.

While results from 3D hydraulic models are expected to be more representative of the physical hydraulic patterns within a stream, it is not yet known what the computational requirements will be for 3D modeling of a complex channel such as the Staunton River. The 2D computational mesh had approximately 100,000 computational nodes per reach, and with the addition of even ten vertical nodes, the resultant model will have to calculate 3D velocity characteristics for over a million nodes, making it likely that only a portion of the reach can be adequately modeled in detail in 3D.

Despite calibration, the relationship between 2D modeled and measured velocity and depth characteristics was much weaker than in other published studies using 2D hydraulic models. There are a number of reasons for this. The first applies to the appropriateness of a 2D model in this highly complex system. Other recently published studies suggested that 2D models did not perform well in the presence of boulders (Waddle 2010) and attributed these errors to errors in topography (Pasternack et al. 2006). However, it is difficult to discern whether these errors illustrate topographic errors or modeling inadequacies (or most likely, both). Three different topographic models were used in this research and the model was highly sensitive to differences in the topographic models. These results indicate that caution be taken in interpreting 2D modeling results in complex streams, and that different topographic model creation scenarios need to be evaluated further.

5.2. Hydraulic Complexity and Spatial Habitat Distribution

While our study was limited to one fish species in one stream, results indicate that metrics based on hydraulic engineering principles may be used to evaluate in-stream habitat at coarse spatial scales (10–30 m) but not without evaluating other nonhydraulic habitat parameters. The use of area-weighted circulation (CRC) indicated there was a threshold value of flow

complexity beyond which, fish density began to decline. More data over a larger area and range of flows would be needed to confirm this hypothesis. While velocity characteristics and flow structure play a role in the brook trout habitat preferences at the local scale the relationships between variables are complex and difficult to predict as with any natural system. This study spanned 11 years and included both spring and autumn samplings providing a unique opportunity to evaluate hydraulic complexity metrics integrated over time at a coarse scale. There was no relationship between conventional weighted usable area habitat characterizations in this study, but when the spatial relationships of high value habitat areas were analyzed, these correlated with age 1+ brook trout biomass and with low velocity and Fr patches. Perhaps due to modeling limitations, the strongest relationship between brook trout populations and habitat were not model results, but the percent protruding boulders, which can be an indicator of both cover and flow complexity.

5.3. *Influence of Urbanization and Riparian Vegetation on Stream Temperature*

Local stream temperature was related to both local scale (riparian vegetation) and watershed scale (imperviousness and watershed area) characteristics. In general, stream temperature was warmer in nonforested reaches than forested reaches and was warmer in urban watersheds than nonurban watersheds. For forested reaches in urban watersheds, the baseline stream temperature was elevated compared to nonurban watersheds, but was cooled and buffered when compared to the paired nonforested urban stream temperatures, illustrating the importance of riparian vegetation in mitigating stream temperature increases from urbanization. This impact was significant at the reach scale (100-200 m) used in this study and seasonal means were independent of which reach (forested or nonforested) was upstream indicating that the influence of riparian vegetation on stream temperature in these mixed land use streams is local and that the overall impact of riparian vegetation on downstream temperatures is unknown. The effect of removing trees from the riparian area was an increase in local maximum summer temperature of 0.7 $^{\circ}$C in nonurban watersheds. The effect of urbanization was an 1.1 $^{\circ}$C increase in local maximum summer temperature, and the combined influence of urbanization and removal of riparian forest was 1.8 $^{\circ}$C in urban watersheds. This simple model provides a means to evaluate

the impact of changes to riparian vegetation or watershed land use on stream temperature. When watershed area is taken into account, the impact of both urbanization and riparian vegetation on stream temperature decreases with increasing watershed size. Summer maximum temperatures increase with increasing watershed size most likely reflecting the decreasing influence of groundwater (baseflow).

Riparian vegetation was also important in decreasing extreme summer temperature frequency and duration. Both maximum daily temperatures and the frequency and duration of elevated temperatures were reduced in forested reaches implying that at this scale, riparian vegetation is important for maintaining temperature regimes suitable for thermally sensitive organisms.

5.4. Future Research

5.4.1. Recommendations for Habitat Modeling

- Model results are highly dependant on representative topography; therefore, to have accurate model results, extreme effort must be undertaken to ensure the most accurate topography.
- Many studies have attributed model errors to errors in topography representation. While the effect of mesh resolution has been fully explored, the need for more detailed topography representation has not.
- To fully evaluate the ability of a 2D or 3D model to represent flow patterns:
    - Many distributed point measurements should be taken, while noting that for very low flows, the error in point measurement could be greater than the model error.
    - A number of cross-sections should be included to evaluate the modeled velocity distribution.
    - Model verification should include a comparison of modeled flow patterns with flow visualizations.

5.4.2. Habitat Analysis for Stream Restoration Design

- While studies are beginning to show the importance of temporally and spatially varying flow to fish habitat, there is still not a good way to link hydraulics to fish habitat at a scale that is relevant to fish populations.
- While this work illustrates methods for quantifying flow complexity and spatial distribution of flow. Future work should investigate the importance of turbulence and 3D flow complexity on fish populations.
- One major source of uncertainty when evaluating hydraulic fish habitat is the method of scaling up from individual fish preferences to quantifying habitat at the reach scale. More work should be done incorporating bioenergetic models with hydraulic characteristics.
- Relationships between spatial organization of habitat at different spatial scales (local, reach, and watershed) should be investigated across a range of flows.

5.4.3. Influence of Urbanization and Riparian Vegetation on Stream Temperature

- To build upon this research evaluating the influence of riparian vegetation and urbanization on stream temperature, climatic variables such as air temperature should be included to model the impact of climate change.
- It is imperative for future stream temperature studies to include the timing, frequency and duration of extreme high temperature events. Recent studies have linked temperature surges following large summer storms to impervious surfaces.

5.5. References

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Appendix A

Matlab Code for CRC, KEG metrics from gridded model output

This code was written to calculate vorticity from x and y velocity vector output from River2D

```
function [x,y,u,v,z,M] =
jessVortDepthM2(fileCSVx,fileCSVy,fileCSVd,fileCSVV,delta)
    %Written by J.P. Resop resop@vt.edu
    %Edited by J.L. Kozarek

    %This function calculates the vorticity at each point in
    %a uniform grid.  There are three input parameters:
    %fileCSVx = The name of the CSV file with x direction flow
    i.e. 'velx.csv'
    %fileCSVx = The name of the CSV file with y direction flow
    %fileCSVd = The name of the CSV file with depth information
    %fileCSVV = The name of the CSV file with velocity
    information
    %delta = The grid cell size
    %[x,y,u,v,z,M] = jessVortDepthM2('velx.csv', 'vely.csv',
    'depth.csv','vel.csv', 0.1)

        tic
    %Load the data from the CSV files and create the data
    matrix
        xmat = load(fileCSVx);
        ymat = load(fileCSVy);
        dmat = load(fileCSVd);
        Vmat = load(fileCSVV);
        data = [xmat(:,2:4) ymat(:,4)];

        dx = delta;
        dy = delta;
        diag = sqrt(2 * delta^2); %Diagonal distance
        temp = size(data);
        n = temp(1);

    %Convert data matrix from column based to a X by Y sized
    matrix
    %First calculate the number of rows and columns
        temp = data(1,2);
        int = 1;
        while (temp == data(int+1,2))
            int = int + 1;
```

```
        end
    intx = int;
    inty = n / int;


%Then create the new matrices for x,y,u, and v
    count = 1;
    for j = 1:inty
        for i = 1:intx
            x(i,j) = data(count,1);
            y(i,j) = data(count,2);
            depth = dmat(count,4);
            if (depth > 0)
                u(i,j) = data(count,3) / depth;
                v(i,j) = data(count,4) / depth;
                Vel(i,j) = Vmat(count,4);
                %Calculate velocity magnitude
                %Vel(i,j) = sqrt(u(i,j)^2 * v(i,j)^2);
            else
                %u(i,j) = 0;
                %v(i,j) = 0;
            end
            count = count + 1;
        end
    end
    z = zeros(intx,inty);

    out(:,1) = xmat(:,1);
    out(:,2) = data(:,1);
    out(:,3) = data(:,2);
    out(:,5) = dmat(:,4);
    out(:,6) = Vmat(:,4);


%Run the numerical method for vorticity
%Warning - Will not calculate vorticity on the edges of the
grid
    sumdiff = 0;
    ignore = 0;
    for j = 2:inty-1
        for i = 2:intx-1
            if ((u(i,j) == 0) & (v(i,j) == 0))
                z(i,j) = 0;
                %ignore = ignore + 1;
            else
                dvdx = (v(i+1,j) - v(i,j)) / dx;
                dudy = (u(i,j+1) - u(i,j)) / dy;
```

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```
                z(i,j) = abs(dvdx - dudy);
                %sumdiff = sumdiff + abs(dvdx - dudy);
            end
            M2 = [0];
            if (Vel(i,j) ~= 0)


%Calculate M2 (KEG) from velocity
foo = 1;
    if (Vel(i-1,j-1) > 0)
        M2(foo) = 2 * mean([Vel(i,j),Vel(i-1,j-1)]) *
        abs((Vel(i,j) - Vel(i-1,j-1))/diag) /
        min(Vel(i,j),Vel(i-1,j-1))^2;
    foo = foo + 1;
    end

    if (Vel(i-1,j) > 0)
        M2(foo) = 2 * mean([Vel(i,j),Vel(i-1,j)]) *
        abs((Vel(i,j) - Vel(i-1,j))/delta) /
        min(Vel(i,j),Vel(i-1,j))^2;
    foo = foo + 1;
    end

    if (Vel(i-1,j+1) > 0)
        M2(foo) = 2 * mean([Vel(i,j),Vel(i-1,j+1)]) *
        abs((Vel(i,j) - Vel(i-1,j+1))/diag) /
        min(Vel(i,j),Vel(i-1,j+1))^2;
    foo = foo + 1;
    end

    if (Vel(i,j-1) > 0)
        M2(foo) = 2 * mean([Vel(i,j),Vel(i,j-1)]) *
        abs((Vel(i,j) - Vel(i,j-1))/delta) /
        min(Vel(i,j),Vel(i,j-1))^2;
    foo = foo + 1;
    end

    if (Vel(i,j+1) > 0)
        M2(foo) = 2 * mean([Vel(i,j),Vel(i,j+1)]) *
        abs((Vel(i,j) - Vel(i,j+1))/delta) /
        min(Vel(i,j),Vel(i,j+1))^2;
    foo = foo + 1;
    end

    if (Vel(i+1,j-1) > 0)
        M2(foo) = 2 * mean([Vel(i,j),Vel(i+1,j-1)]) *
        abs((Vel(i,j) - Vel(i+1,j-1))/diag) /
```

```
        min(Vel(i,j),Vel(i+1,j-1))^2;
foo = foo + 1;
end

if (Vel(i+1,j) > 0)
    M2(foo) = 2 * mean([Vel(i,j),Vel(i+1,j)]) *
    abs((Vel(i,j) - Vel(i+1,j))/delta) /
    min(Vel(i,j),Vel(i+1,j))^2;
foo = foo + 1;
end

if (Vel(i+1,j+1) > 0)
    M2(foo) = 2 * mean([Vel(i,j),Vel(i+1,j+1)]) *
    abs((Vel(i,j) - Vel(i+1,j+1))/diag) /
    min(Vel(i,j),Vel(i+1,j+1))^2;
foo = foo + 1;
end
end

            M(i,j)= mean(M2);
            %out((j-1)*intx+i,1) = x(i,j);
            %out((j-1)*intx+i,2) = y(i,j);
            out((j-1)*intx+i,4) = z(i,j);
            out((j-1)*intx+i,7) = mean(M2);
        end
        j / (inty-1) * 100
        toc
    end

    header = ['n','x','y','T','D','V','K'];
    dlmwrite('output.csv',header,',');
    dlmwrite('output.csv',out,'-append');

        toc

end
```