DISSERTATION

EVALUATING THE IMPACTS OF URBANIZATION AND STORMWATER MANAGEMENT PRACTICES ON STREAM RESPONSE

Submitted by

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In partial fulfillment of the requirements

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WE HEREBY RECOMMEND THAT THE DISSERTATION PREPARED UNDER OUR SUPERVISION BY CHRISTINE A. POMEROY ENTITLED "EVALUATING THE IMPACTS OF URBANIZATION AND STORMWATER MANAGEMENT PRACTICES ON STREAM RESPONSE" BE ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY.

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ABSTRACT OF DISSERTATION

EVALUATING THE IMPACTS OF URBANIZATION AND STORMWATER MANAGEMENT PRACTICES ON STREAM RESPONSE

Land use changes related to urbanization can have profound impacts on runoff characteristics, resulting in changes to the magnitude, frequency, and duration of flow in stream channels. These flow regime changes often alter the quality of aquatic habitats and native biota of streams. Identification of linkages between urban land use policies and practices and the associated geomorphic and ecological consequences in urban streams is needed when evaluating the effectiveness of urban stormwater runoff management practices, including the management of urban development and limiting percent of impervious surface cover.

In this study, protocols were developed to help standardize data generation for identifying mechanistic linkages to provide public agencies with tools to identify management practices that will achieve the fewest ecological impacts and sustain physical habitats and ecological conditions in urban streams. The protocols were applied in the North Carolina Piedmont. Eight watersheds spanning a gradient of urbanization from undeveloped to highly developed were used to identify linkages between indicators of aquatic ecosystem health and hydrologic and geomorphic metrics derived from a 20-year continuous stream flow record. The 20-year continuous stream flow records for each

watershed were generated using EPA Storm Water Management (SWMM5) models calibrated to 18 months of measured flow data.

The macroinvertebrate metrics Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness, EPT percent richness, and a benthic index of biotic integrity (B-IBI) for the eight North Carolina Piedmont watersheds were found to be linearly related to the $T_{0.5}$ hydrologic metric computed for 1-, 2-, 5-, 10-, and 20-year time periods. Qualitative multihabitat EPT richness metric values were strongly related to the $T_{0.5}$ hydrologic metric, which describes the percent of time discharge is above the peak discharge of the 0.5-year return interval storm. The $T_{0.5}$ values computed from the 1- and 2-, and 10-year time periods were the best predictors of macroinvertebrate metric scores. $T_{0.5}$ values computed from both 15-minute and hourly time steps were good predictors of macroinvertebrate metric values; $T_{0.5}$ values computed from daily time steps were poor predictors of macroinvertebrate response.

The T_{Qmean} hydrologic metric (percent of time discharge is above the mean annual discharge value) was not a good predictor of macroinvertebrate response, and the T_1 hydrologic metric (percent of time discharge is above the peak discharge of the 1-year return interval storm) did not predict macroinvertebrate response as well as the $T_{0.5}$ hydrologic metric. The T_2 hydrologic metric (percent of time discharge is above the peak discharge of the 2-year return interval storm) was a better predictor of macroinvertebrate response than both the $T_{0.5}$ and the T_1 hydrologic metrics. T_2 values computed from an annual maximum series, rather than a partial duration series, were better predictors of macroinvertebrate response for the longer (10- and 20-year) time periods, but offered approximately the same level of prediction when computed for the 5-year time period.

Macroinvertebrate metrics were shown to be logarithmically related to predominant substrate size, with EPT richness for the richest targeted habitat having the strongest correlation. Predominant grain size was better predictor of macroinvertebrate metrics than cumulative excess shear stress, except for EPT richness for the qualitative multi-habitat results. Similar relationships were observed between the macroinvertebrate metrics and the cumulative excess shear stress computed for the 1-, 2-, 5-, 10-, and 20-year time periods.

The $T_{0.5}$ hydrologic metric was affected by both land use and channel geometry. When land use in a watershed was changed from rural to urban, but the channel geometry remained the same, the value of the $T_{0.5}$ hydrologic metric did not decrease as much as when both land use and channel geometry were changed. When the channel geometry of the streams modeled for the undeveloped Morgan Creek watershed were changed to be similar to those observed in urban watersheds of the North Carolina Piedmont, but land use remained the same (rural), the $T_{0.5}$ hydrologic metric was nearly the same as the one computed when both land use and channel geometry were changed. This indicates that channel geometry plays a significant role in values of the $T_{0.5}$ hydrologic metric.

Modification of rural channel cross-sections to be similar to cross-sections observed in urban watersheds of similar size changed the $T_{0.5}$ hydrologic metric enough to cause it to predict a degraded EPT richness score from good-fair or fair, to poor. When stormwater controls were applied to the developed Morgan Creek watershed, and when channel geometry was left in its original state, changes in the value of the $T_{0.5}$ hydrologic metric were minimized, and anticipated changes to macroinvertebrate ratings were reduced. When stormwater controls were applied to a developed Morgan Creek

watershed, but channel geometry was changed to be similar to cross-sections observed in urban watersheds, the $T_{0.5}$ hydrologic metric score was greatly improved, but was not improved enough to minimize changes in the macroinvertebrate score, which was almost always reduced to fair. Long-term changes in erosion potential in the Morgan Creek watershed were minimized when stormwater controls were applied.

Biotic response was shown to be related to both hydrologic or geomorphic metric values, indicating that restoration of hydrology may not allow for the attainment of biotic goals if habitat is not also restored. In urbanizing situations, preservation of a more natural hydrology and restrictions on increases in erosion potential will reduce geomorphic changes from what would occur if stormwater management controls were not implemented, although changes in upland sediment supply also must be accounted for. Control of channel erosion was shown to be important because channel geometry was found to directly influence hydrologic metric values due to its impact on conveyance capabilities and loss of flood attenuation due to disconnectivity from floodplains. These relationships were shown for the North Carolina Piedmont, but are also applicable throughout the United States.

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DEDICATION

For Eva
whose spirit is always with me as she
continues to make her mark on the world.

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1.0 INTRODUCTION

1.1 Background for this Study

Land use change, especially urbanization, has profound impacts on the runoff characteristics of land that it occupies and, consequently, on the aquatic environments of the urban streams to which that runoff drains (WEF and ASCE, 1998). The fundamental hydrologic changes that are associated with urbanization include more frequent and higher magnitude flows, increased duration of geomorphically significant flows, flashier/less predictable flows, altered timing and rate of change relative to riparian and floodplain connections, altered duration of low-flow periods, and conversion of subsurface distributed discharge inputs to surface discharges (Roesner and Bledsoe, 2003). All of these observed phenomena are interrelated.

It was noted by Roesner and Bledsoe (2003) that linking stream characteristics to gross measures of development, such as total imperviousness, percent agriculture, or human population, provides little meaningful information for understanding key processes and creating practical mitigation strategies. In the past, studies have typically referred to the influences of urbanization in terms of percent impervious or percent land cover, without specifically addressing land use types or stormwater management practices such as conveyance characteristics, (e.g. pipes or swales), the use of detention, or best management practices (BMPs). The dominant trend in these studies has been to use simple linear regressions to link gross percentage of impervious area in a watershed with biological populations (Maxted and Shaver, 1996; Center for Watershed Protection 2002; Roesner and Bledsoe 2003). In order for the specific mechanisms through which urbanization degrades aquatic ecosystems to be identified and understood, accountability of hydrologic and geomorphic changes, land cover, and drainage system hydraulic features is needed (Roesner and Bledsoe 2003).

Based on the findings and recommendations of the study conducted by Roesner and Bledsoe (2003) the Water Environment Research Foundation (WERF) funded a research study to develop a protocol that municipalities responsible for stormwater management could use to evaluate the impacts of land use patterns and alternative stormwater management strategies on streams in urbanizing watersheds. The study called for the development of the protocols followed by a test application of the protocols to a real watershed. This research project formed the basis for this dissertation.

1.2 Objectives of Research

The research described in this dissertation was funded by the Water Environment Research Foundation to meet the following objectives:

- Develop study protocols and diagnostic measures to help standardize data generation for identifying the linkages between urban land use policies and practices, stormwater runoff characteristics, geomorphic parameters, and effects on aquatic habitat and biota.
- Apply the protocols using a case study to verify the accuracy and usefulness of the protocols and diagnostic measures.

The study protocols were developed to evaluate the impacts of land use patterns and alternative stormwater management strategies on the biotic integrity and geomorphic stability of streams in urbanizing watersheds, and included:

- creating a "big picture" protocol to describe the overall approach,
- providing guidelines on data needed to implement protocols and methods to generate data not obtained in the field (i.e., the generation of stream flow records using rainfall/runoff models), and
- identifying tools that can be used to evaluate the impacts of alternative urbanization strategies on biotic integrity and geomorphic stability

1.3 Main Hypotheses of Research

The main hypothesis of this research is that hydrologic and geomorphic metrics can be used to establish linkages between stormwater management practices (including land use and runoff controls) and stream ecosystem health as measured by macroinvertebrates. These metrics are computed from continuous, long-term, instream flow records. Establishment of these linkages can allow municipalities to relate land use planning and runoff control strategies to biotic health in streams, and thus identify stormwater management criteria that will provide protection of urban streams. This relationship is illustrated pictorially in Figure 1-1.

A corollary hypothesis is that rainfall/runoff models can be used to generate the stream flow data needed for the computation of hydrologic and geomorphic metrics for the purposes of evaluating the effects of stormwater management practices. While instream flow monitoring has been conducted extensively across the United States by the

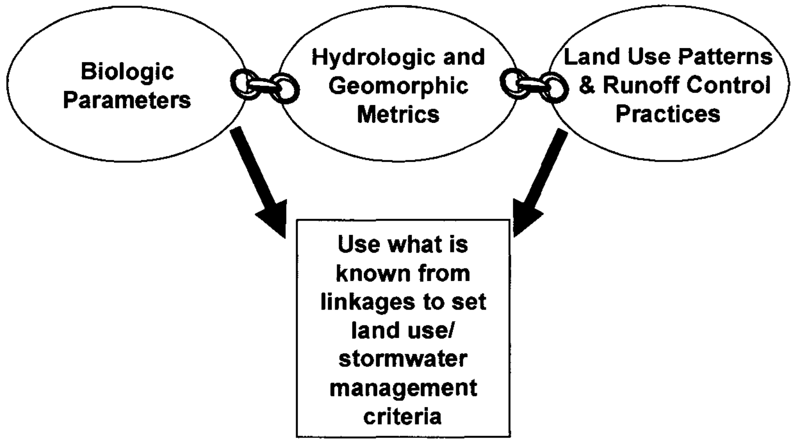


Figure 1-1. Linking Land Use Patterns and Runoff Control Practices to Stream Biologic Integrity Through Hydrologic and Geomorphic Metrics.

USGS and other entities, long-term flow monitoring data are not readily available for watersheds at a scale that allows for the evaluation of the effect of stormwater management practices on hydrologic and geomorphic metrics. The effects of stormwater management practices on these metrics are best evaluated in watersheds where the management practices are applied throughout the entire watershed, as downstream effects can be easily masked by uncontrolled areas. Lack of sufficient monitoring data necessitates the use of rainfall/runoff models to generate the stream flow data needed for the computation of hydrologic and geomorphic metrics in linkage development.

If these hypotheses are true, then a framework protocol can be developed that assists municipalities in gathering the data required and performing the analysis necessary to determine which stormwater management criteria are most protective of receiving stream environments in their region. In addition to testing of the hypotheses described above, this dissertation describes the protocol framework and provides examples of applications and testing of the protocol.

1.4 Testing the Protocol

Individual hydrologic and geomorphic components of the protocol were tested using watersheds in Fort Collins, Colorado and Lenexa, Kansas. Hydrologic and hydraulic modeling with EPA SWMM4.4h was used to evaluate the effectiveness of typical stormwater management practices on hydrologic and geomorphic metrics in a small watershed (30 acres) in Fort Collins, Colorado (Rohrer and Roesner, 2006; Edgerly, 2006; Edgerly et. al., 2006; Pomeroy and Roesner, in press). In Lenexa, Kansas portions of the protocol were applied to develop design criteria for subdivision level detention basins so that the current geomorphic condition in the natural stream system would be preserved to the maximum extent practicable (Rohrer et al., 2005; Pomeroy and Roesner, *in press*; Pomeroy et al., *in press* a). Three peer-reviewed publications were generated from these analyses: Rohrer and Roesner, 2006; Pomeroy and Roesner, *in press*; Pomeroy et al., *in press* a.

Watersheds in the North Carolina Piedmont (Raleigh-Durham metropolitan area) were used to test linkages between land use change, stormwater runoff characteristics, geomorphic parameters, and effects on benthic macroinvertebrates. SWMM5 model-generated stream flow data as well as data gathered by the USGS under the National Water Quality Assessment (NAWQA) Program study Effects of Urbanization on Stream Ecosystems (USGS, 2007) were used to evaluate linkages in eight watersheds.

1.5 Organization of Dissertation

Chapter 2.0 provides an overview of recent literature that describes linkages between urbanization and environmental parameters. Chapter 3.0 addresses the protocol for linking urbanization to water environment parameters; i.e., hydrologic, geomorphic, and biologic metrics. Chapter 3.0 also includes discussion of preliminary testing of individual hydrologic and geomorphic components of the protocol in Fort Collins, Colorado and Lenexa, Kansas. Chapter 4.0 describes the development and calibration of the models for the application of the protocol in the North Carolina Piedmont. Chapter 5.0 shows the relationship of hydrologic and geomorphic metrics to land use changes and runoff controls in the North Carolina Piedmont application. Chapter 6.0 includes a summary of findings, conclusions and recommendations for future research.

2.0 LITERATURE REVIEW

2.1 Introduction

Review of recent literature reveals a number of advancements in research regarding the linkages between urbanization, hydrology, hydraulics, geomorphology, and the biotic integrity of streams. This review covers three major areas of research that are important to the establishment of these linkages: 1) ecosystem flow and habitat requirements, 2) developments in refining urbanization-aquatic ecosystem linkages; and 3) mechanisms of altered flow regimes.

2.2 Ecosystem Flow and Habitat Requirements

Maddock et al. (2004) discussed the complex interaction of four primary factors in determining habitat in aquatic systems: water quality, the energy budget, the physical structure of the channel, and the flow regime. The combination of the latter two factors results in the instream physical habitat for biota in lotic ecosystems. In addition to these four factors, sediment sources and dynamics, combined with the physical structure of the channel and flow regime determine habitat. Sediment dynamics are determined in part by hydraulics, but are also determined by a combination of watershed and local geologic settings, local channel boundary conditions, and stream network position (Fitzpatrick and Peppler, 2007). With regard to flow regime, a major trend in research on determining the optimal flows is approaching the problem from the bottom-up, one species at a time. Studies on the flow preferences of individual species have been conducted for decades (Statzner and Moss, 2004).

Shear stress has also recently gained attention in the ecohydraulics community (Tharme, 2003). A hydrologic change that increased shear stress or reduced the thickness of the viscous sublayer would reduce the availability of suitable microhabitats for some species, while increasing habitat availability for other species, which alters community functioning and food-web dynamics (Gore et al., 2001). Merigoux and Doledec (2004) also examined the relationship between near-bed hydraulics and benthic invertebrate response. With direct measurements of near-bed hydraulic conditions and samples of invertebrate taxa distribution, Merigoux and Doledec (2004) were able to show relationships between hydraulic parameters and taxon richness, biological traits, and functional diversity using simple linear regressions. Shear stress and Froude number were found to be the most important hydraulic parameters, while substrate particle size and bed roughness appeared to be less influential (Merigoux and Doledec, 2004).

In addition to considerations of the hydraulic component of physical stream habitat, there is a substantial body of literature that provides anecdotal, but important links between aquatic species and specific physical habitat features and more recently, quantitative information linking physical features to the presence and abundance of various aquatic taxa (e.g., Kaufmann et al., 1999). Maddock (1999) pointed out that key research issues in physical stream habitat evaluation lie in decisions regarding the appropriate level of detail providing a meaningful, yet cost-effective investigation, as well as identifying which factors are truly biologically relevant rather than simply geomorphic features. The United States Environmental Protection Agency's (U.S. EPA) Environmental Monitoring and Assessment Program (EMAP) has been a driving force in obtaining much quantitative habitat information and recently has performed studies in both the Western and Eastern United States. When planning for the EMAP physical data collection effort, U.S. EPA met with experts in these areas and developed a list of seven general categories of physical habitat attributes thought to have important influences on the biota of streams (Kaufmann, 1993). These included:

- Stream Size - Channel Dimensions
- Channel Gradient
- Channel Substrate Size and Type
- Habitat Complexity and Cover
- Riparian Vegetation Cover and Structure
- Anthropogenic Alterations
- Channel-Riparian Interaction

These are similar to the categories used in Ohio EPA's Qualitative Habitat Evaluation Index (QHEI) (Rankin, 1989; 1995). U.S. EPA EMAP efforts have contributed to the recent focus on obtaining data linking these physical features to aquatic community level features in a quantitative manner. Often the biological or habitat data have been either relatively weak or not quantified in a repeatable fashion. Data collected over the last 10 years are beginning to produce quantitative data that should add accuracy in identifying gradients of biological response to various physical habitat features.

Recent habitat modeling methodologies are commonly based on the integration of hydrologic, hydraulic, and biological response data such as model depth, velocity, substratum, cover, and benthic shear stress. These data are used to develop habitat-discharge curves for biota, which are then used to predict optimum flows (Tharme, 2003). Current modeling approaches addressing stream/river flow regimes and their effects on habitat and associated biota have their basis in fisheries restoration efforts in flow-regulated rivers. Probably the most widely used method for analyzing linkages between flow and physical habitat for the purpose of formulating management options is the Instream Flow Incremental Methodology (IFIM), designed by the U.S. Fish and Wildlife Service (USFWS) for river system management (Bovee et al., 1998) and the associated Physical Habitat Simulation (PHABSIM) software (Milhous, 1984). PHABSIM is an extensive set of programs designed to predict the potential micro-habitat conditions in rivers as a function of discharge and the relative suitability of those conditions to aquatic life based on empirical preference curves (Maddock, 1999). An underlying assumption of the methodology is that fish population dynamics are directly related to habitat availability, and that habitat availability is directly related to a river's flow regime. It is debatable as to whether biota use habitat in proportion to its availability because biotic interactions may be the ultimate determinant of habitat use (Jacobson et al., 2001).

In a review of applications of the IFIM and PHABSIM over the past 20 years, Gore et al. (2001) identified five hydraulic conditions that most affect the distribution and ecological success of lotic biota: suspended load, bedload movement, water column effects (e.g., turbulence), velocity profile, and near-bed hydraulics. Their review noted that differences between predicted optimal and minimum flow criteria suggest that basing a management decision on a few target invertebrate species may result in the loss of habitat for other non-target species. A larger proportion of benthic species can be provided habitat by using flows that are appropriate for functional feeding group targets or for highest community diversity. If the targets of high community diversity among benthic macroinvertebrates or the appropriate proportion of functional feeding groups are met, it will likely ensure a diversity of hydraulic conditions within the stream reach of interest. Producing suitability criteria for highest levels of benthic community diversity is a relatively simple procedure (Gore et al., 2001).

PHABSIM has also been used to examine linkages between flow and numerous variables affecting aquatic habitat and biota. Wu and Wang (2002) examined the relationship between substrate variation and flow in the framework of physical habitat simulation. The results implied that flow-related habitat studies are not usually sensitive to substrate alteration. Osmundson et al. (2002) demonstrate how flow regime can affect multiple aspects of an aquatic ecosystem through an indirect chain of linkages. They established statistical relationships between extended duration of fine sediment accumulation due to loss of flushing flow by a dam, depressed macrophyte production, reduced macroinvertebrate community integrity, and the health and survival of the pike minnow. Maddock et al. (2004) investigated the impact of channel morphology and flow on physical habitat availability for the two-spined blackfish (Gadopsis bispinosus) using PHABSIM. They found that the configuration of the channel bed topography is influential for the hydraulic habitat present at a given discharge and the response of this hydraulic habitat to discharge.

Building upon an established understanding of hydraulic habitat requirements of target fish species, Maddock et al. (2004) determined that streams with varying channel bed topography provide wide ranging usable hydraulic habitat for the target species under the range of modeled flows. Excessively high water velocities were identified as the hydraulic habitat feature most limiting success of the target species; therefore, research and modeling efforts were able to provide insight to physical stream habitat attributes limiting populations of the target species. Similarly, Booker and Dunbar (2004) utilized PHABSIM modeling to determine that greater suitable physical stream habitat was available over a wider range of hydraulic conditions in more natural river channels than in more engineered channels, thus contributing to the understanding of the interactions among physical channel characteristics, hydraulic conditions, and sustainable ecological communities in urban streams. Lamoroux et al. (1999), working in the Rhone River basin in France, sought to evaluate how hydraulic habitat affects fish communities. They concluded that hydraulics have a stronger affect on fish community structure within given geographic regions than if data for multiple geographic regions are considered together.

Thomson et al. (2004) also evaluated inter-regional differences in habitat-biota relationships when they examined the ecological significance of a geomorphic river characterization scheme by comparing macroinvertebrate assemblages and habitat characteristics of pool and run geomorphic units in three different types of rivers. The study tested the hypothesis that if geomorphic units are physically similar, they should also exhibit some biological similarities. Their results showed significant correlation between variation in macroinvertebrate data and several local habitat variables across river types, suggesting that local factors are more important than regional factors (Thomson et al., 2004).

Other researchers such as Lammert and Allan (1999) have evaluated habitat-biota relationships for hydraulic conditions along with other physical habitat characteristics at multiple spatial scales in an attempt to evaluate the relative importance of these factors in determining lotic communities. In a southeastern Michigan study, they observed that fish community composition showed a stronger relationship to flow variability and adjacent land use conditions, whereas macroinvertebrates were more strongly correlated with dominant substrate characteristics. More recently, Townsend et al. (2004) compared streams in pastureland and control streams in undeveloped, non-agricultural catchments with regard to macroinvertebrate community composition, as well as to a number of physical channel characteristics, hydraulic variables, and physical habitat characteristics. Significant differences were observed between the two stream types with regard to water velocities, physical habitat conditions, and macroinvertebrate community composition and condition, though no conclusions were drawn regarding physical factors as determinants of the macroinvertebrate community in the study streams.

2.3 Developments in Refining Urbanization-Aquatic Ecosystem Linkages

It was noted in Roesner and Bledsoe (2003) that linking stream characteristics to gross measures of development, such as total imperviousness, percent agriculture, or human population, provides little meaningful information for understanding key processes and creating practical mitigation strategies. In the past, studies have typically referred to the influences of urbanization in terms of percent impervious or percent land cover, without specifically addressing land use types or stormwater management practices such as conveyance characteristics, (e.g., pipes or swales), the use of detention, or best management practices (BMPs). The dominant trend in these studies was to use simple linear regressions to link gross percentage of impervious area in a watershed with biological populations (Center for Watershed Protection, 2002; Roesner and Bledsoe, 2003). In order for the specific mechanisms, through which urbanization degrades aquatic ecosystems, to be identified and understood, accountability of hydrologic and geomorphic changes, land cover, and drainage system hydraulic features is needed (Roesner and Bledsoe, 2003).

A few recent studies have acknowledged the weakness of the common gross percentage impervious area as a predictor of aquatic community health and have attempted to better characterize changes in land cover associated with urbanization in hope of finding more direct linkages to observed effects (Alberti et al., 2003; Booth et al., 2004; Walsh, 2004; Walsh et al., 2004). Booth et al. (2004) point out that the common definition of total impervious area (TIA) is hydrologically incomplete because it may overestimate pervious area in some cases by failing to account for compaction of soils, and underestimate pervious area in other cases by failing to account for infiltration of runoff from small, isolated impervious surfaces that drain to adjacent pervious surfaces.

Total impervious area is not always an accurate predictor of effective impervious area. The degree to which impervious areas are connected to each other and ultimately to the stream is an important factor to consider when characterizing urban development (Booth and Jackson, 1997; Alberti et al., 2003; Walsh, 2004). Direct measurement of directly connected impervious area (DCIA) is complicated (Booth and Jackson 1997), however, and most studies do not differentiate between TIA and DCIA (Brabec et al., 2002). Lee and Heaney (2003) demonstrated the importance of distinguishing between TIA and DCIA when analyzing the hydrology of a system and showed that commonly used remote sensing techniques have limited accuracy for estimating DCIA. In the first part of their study, they used hourly long-term rainfall data and detailed estimates of land cover and hydraulic connection of surfaces in an apartment complex in Miami, Florida, to develop a linear rainfall-runoff estimation model. They found that DCIA made up 44% of the TIA, but contributed 72% of the total runoff volume over 52 years. The second part of their study compared five levels of accuracy in estimation of imperviousness for a development in Boulder, Colorado. The five imperviousness estimation methods with increasing levels of accuracy are as follows: 1) literature reference data; 2) estimating TIA from a GIS database; 3) using a GIS database and subtracting isolated impervious areas from TIA to estimate DCIA; 4) using a GIS database and field investigation to include curbed streets in the DCIA estimate; 5) using a GIS database and field investigation for the entire area including rooftops with downspouts to estimate DCIA. The resulting estimates of DCIA obtained from these methods decreased significantly with increasing accuracy. As a result, runoff estimates generated by the SWMM4.4 model using the values of imperviousness with the five levels of accuracy showed a significant decrease in the total volume of runoff predicted as accuracy of DCIA estimation increased. Lee and Heaney (2003) also found that transportation related impervious area comprised 64% of the TIA and 97.2% of DCIA. This result confirmed the findings of Alberti et al. (2003), who used multivariate analysis to show that road intensity is a good predictor of the spatial arrangement of impervious surfaces.

Using geographic information systems (GIS), Alberti et al. (2003) assessed the usefulness of urban pattern metrics for describing the spatial distributions, configuration, and the hydraulic connectivity of landscape features (e.g., road density, number of road crossings per kilometer of stream, degree of fragmentation, etc.) as predictors of macroinvertebrate community health measured as the benthic index of biotic integrity (B-IBI). They found that road density and number of road crossings were better predictors of B-IBI scores than total impervious area. They also found that the spatial arrangement of development patterns, particularly the amount and configuration of impervious surfaces and forest patches, plays an important role in determining the impact on stream macroinvertebrate communities and could explain much of the variability in B-IBI not explained by TIA. The results of their scale analysis suggest that basin-scale landscape configuration may overshadow the efficacy of local-scale riparian function.

Walsh (2004) measured macroinvertebrate assemblage composition across an urban density gradient and assessed whether the variance could be explained by percent impervious area, drainage connection, septic tank density, catchment area, elevation, and geographic location. With respect to percent imperviousness, the typical patterns were observed: low catchment imperviousness was associated with diverse assemblages and high imperviousness was associated with disturbance-tolerant taxa. In the low to moderate range of imperviousness, however, there was a wide variation in assemblage composition and richness, for which the strongest independent correlate was piped drainage connection. These results suggest that direct measurement of drainage connection is critical when estimating effective imperviousness. Walsh also recommends that efforts to restore streams in urban catchments should start with attention to the catchment drainage system. He suggests that conflicting reports among studies of dominant scale effects (local land use vs. catchment scale land use) could be explained by failure to account for differences in drainage connection.

In response to implications by researchers that riparian buffers were possibly the most important elements in protecting streams from the effects of urbanization (Horner et al., 2001), Walsh (2004) hypothesized that stormwater drainage system disconnect is of greater importance to aquatic biota than riparian buffers. In a study of the effects of urbanization on the geomorphology, habitat, hydrology and fish index of biotic integrity of streams in the Chicago area, Illinois and Wisconsin, Fitzpatrick et al. (2005) showed that the presence of stream buffers with forest and wetlands did not influence geomorphic, habitat, or hydrologic characteristics, or IBI scores. This suggests that forested riparian buffers are less able to moderate the influence of urbanization on biotic integrity of streams probably because of hydrologic alterations such as storm sewers, detention basins, and point-source discharges that directly bypass riparian buffers.

A number of researchers are now advocating low impact development (LID) as an ecologically sound option for stormwater management, although most stress the fact that these new techniques do not eliminate the need for conventional, centralized stormwater management practices, such as detention (Konrad and Burges, 2001; May and Horner, 2002; Shaver and Ridley, 2002; Coffman, 2003; Holman-Dodds et al., 2003; Kirby, 2003). LID concepts for new development incorporate minimizing imperviousness, conserving native vegetation and soil, and implementing integrated, on-site treatment of stormwater (May and Horner, 2002). The theoretical advantage of such systems over a centralized stormwater management approach is that they manage the flow before it is concentrated in the drainage system, using spatially diffuse storage and release, a situation that is closer to the natural, pre-development runoff processes (Konrad and Burges, 2001).

One of the design principles for LID is clustered, compact development patterns, based on the assumption that compact urban forms, (smaller lot sizes and higher density), are more ecologically sound because they preserve open-space (May and Horner, 2002; Booth et al., 2002; Alberti et al., 2003). There is little hard evidence, however, to support this assumption (Alberti et al., 2003; Center for Watershed Protection, 2003). Alberti et al. (2003) show strong statistical relationships between an increase of the percentage of impervious surfaces, their aggregation, and ecological conditions in streams. In particular, as the probability of adjacency of paved areas rises from 50% to 100%, typical B-IBI values decline from 50 (excellent) to 10 (very poor). While the findings clearly suggest that patterns of urban development impact watershed function, this relationship does not indicate a specific threshold. To further confound the issue, it was found that landscape configuration metrics (describing the degree of scatter and the proximity to the stream) are better predictors of B-IBI at the basin scale, while landscape composition metrics (i.e., percent TIA and percent trees) are better at the local scale. This supports the finding that scale is an important factor in assessing the effects of landscape patterns on ecological conditions.

2.4 Mechanisms of Altered Flow Regimes

After having discussed how urbanization changes the hydrologically significant aspects of land cover, a logical next step is to take a closer look at the characteristics of flow regimes, how flow regimes are altered by urbanization, and the ecological ramifications of these alterations. Hydrologic metrics that demonstrate altered streamflow regimes can provide a direct mechanistic link between aspects of urban development and degraded stream ecosystems (Booth et al., 2004).

Flow is a major driver behind many processes that ultimately determine the state of the stream ecosystem (Poff et al., 1997; Poff and Ward, 1989; Bunn and Arthington, 2002; Nilsson and Svedmark, 2002; Riis and Biggs, 2003; Roesner and Bledsoe, 2003), however, it can be very difficult to determine which mechanism is directly responsible for observed impacts in a stream (Bunn and Arthington, 2002). A major strength of a management plan based on the flow regime is that it has a reduced risk of overlooking flow requirements for important components of the ecosystem (Richter et al., 1996; Poff et al., 1997; Tharme et al., 2003; Suen et al., 2004). The five aspects of a flow regime identified as most important to the geomorphology, physical habitat, and ultimately stream biota are: magnitude, frequency, duration, timing, and rate of change (Poff et al., 1997).

The recognition of the importance of flow regime to the ecological integrity of lotic systems has prompted a number of researchers to recommend measures for restoring stream hydrology to a condition more closely approximating the natural flow regime, as part of overall ecological restoration efforts (e.g., Erskine, 1999; Ward et al., 2002; Poff et al., 1997; Richter et al., 1997, 1998; Richter and Richter, 2000; Richter et al., 2003).

Though these recommendations are often directed toward restoration of natural flow regimes following dam construction and associated flow regulation, the principles underlying the recommendations are applicable to streams in urbanizing watersheds where flow regimes have been altered due to development. Management efforts involving flow regime restoration are more tractable for regulated rivers, for which the stream discharge may be more readily manipulated. For streams in urbanizing watersheds, however, the matter of flow regime stabilization or restoration is a more complex matter, involving runoff and flow mitigation in response to changes in land cover and drainage patterns.

One of the most widely used techniques for quantifying changes in flow regime caused by human activity is the Indicators of Hydrologic Alteration (IHA) method developed by Richter et al. (1996) and the Range of Variability Approach (RVA), which is a modification of the IHA (Richter et al., 1997). The IHA uses continuous stream flow data (either from stream gauges or model-generated) to generate 32 hydrologic metrics characterizing the ecologically significant features of the flow regime. Hydrologic perturbations are evaluated by comparing "pre-impact" and "post-impact" time frames, using the central tendency and dispersion of each parameter to generate 64 Indicators of Hydrologic Alteration (IHA). The IHA statistics consist of five groups:

- magnitude of monthly water conditions;
- magnitude and duration of annual extreme water conditions;
- timing of annual extreme water conditions such as annual maxima and minima one-day means or 30-day means;

- frequency and duration of high (above the 75th percentile) and low (below the 25th percentile) pulses; and rate and frequency of water condition changes.

A summary of the hydrologic parameters used in the IHA and their characteristics is given in Table 2-1. It is recommended that at least 20 years of record be used to differentiate between natural variation and regime alteration. In the RVA, the 32 hydrological metrics from the IHA method (Richter et al., 1996) are computed for every year of an unregulated daily streamflow record, characterizing: monthly flow variations, magnitude, frequency, timing of extreme events, and rise and fall rates of the hydrograph.

Table 2-1. Summary of Hydrologic Parameters Used in the IHA and Their Characteristics.

IHA Statistics	Regime Characteristics	Hydrologic Parameters
Group 1		
Magnitude of monthly water conditions	Magnitude, Timing	Mean value for each calendar month
Group 2		
Magnitude and duration of annual extreme water conditions	Magnitude, Duration	Annual maxima 1-day means Annual minima 1-day means Annual maxima 3-day means Annual minima 3-day means Annual maxima 7-day means Annual minima 7-day means Annual maxima 30-day means Annual minima 30-day means Annual maxima 90-day means Annual minima 90-day means
Group 3		
Timing of annual extreme water conditions	Timing	Julian date of each annual 1-day maximum Julian date of each annual 1-day minimum
Group 4		
Frequency and duration of high and low pulses	Magnitude, Frequency, Duration	No. of high pulses each year No. of low pulses each year Mean duration of high pulses within each year Mean duration of low pulses within each year
Group 5		
Rage and frequency of water condition changes	Frequency, Rate of change	Means of all positive differences between consecutive daily means Means of all negative differences between consecutive daily values No. of rises No. of falls

Source: Richter et al. (1996)

The natural annual range of these metrics is identified and used to set acceptable ranges of flow to be used as management targets such as plus or minus one standard deviation.

Suen et al. (2004) point out that a practical drawback of the IHA is that it is impossible to optimize all 32 metrics. As a result of this realization, there have been some attempts to reduce the number of indices needed from the IHA by identifying a subset of key metrics that are most ecologically important or that avoid redundancy in the information they provide (Clausen and Biggs, 1997; 2000; Olden and Poff, 2003).

Stewardson and Gippel (2003) point out that a limitation of the RVA approach proposed by Richter et al. (1996; 1997; 1998) is that there is no specific way of relating ecologic response to the flow statistics used. They propose the Events Flow Method (EFM) as a way to develop flow targets by explicitly identifying geomorphic and biological processes affected by flow variability. This is done by selecting threshold discharges, event magnitudes and durations for events based on expert knowledge of important time-scales for geomorphic and biological processes.

Clausen and Biggs (1997) attempted to determine the most ecologically relevant hydrologic indices for characterizing flow regimes by computing 34 hydrologic variables from daily flow records at 83 sites and testing them for statistically significant relationships with periphyton and invertebrates. They found that the frequency of flows greater than three times the median flow was the most useful in explaining the ecologic condition because it explained four out of six main benthic community measures. In a later study, Clausen and Biggs (2000) calculated 35 flow variables for 62 perennial rivers from seven-year flow records of the same time period and identified four groups of variables based on covariance among sites using a principal component analysis, PCA.

The highly correlated groups of variables were: 1) size of river (average flow magnitude); 2) overall variability of flow and magnitude of extreme events (intensity of disturbance); 3) volume and duration of high flows (bed movement and degree of disturbance); and 4) the frequency of high flow events.

Olden and Poff (2003) reviewed 171 hydrologic indices using long-term continuous flow records from 420 sites and used a PCA along with ecologically based recommendations to select a reduced set of indices that sufficiently represent the critical attributes of the flow regime. The ecologically based recommendations included evaluation of ecologically important streamflow characteristics that constitute the natural flow regime, including the seasonal patterning of flows; timing of extreme flows; the frequency, predictability, and duration of floods, droughts, and intermittent flows; daily, seasonal, and annual flow variability; and rates of change (Poff et al., 1997). Olden and Poff (2003) recommend using a subset of two to four indices from 171 indices to adequately represent the biologically critical portions of the flow regime. They also examined whether hydrologic indices were transferable among different "stream types." The six different "stream types" were chosen to cover a range of regional flow regime types, including: harsh intermittent; intermittent flashy; snowmelt; snow and rain; superstable or stable groundwater; and perennial flashy. They found that indices explaining dominant patterns of variance were stream-type specific, demonstrating that hydroclimatic characteristics of a study region should be considered when choosing hydrologic indices for a study.

McMahon et al. (2003) quantified the difference between flow-based and stage-based metrics in urban and non-urban catchments. The primary motivation for this comparison was that if stage-based metrics were found to be comparable to the widely used flow-based metrics, they could provide a cost-effective alternative hydrologic measure in catchments lacking flow-based metrics. They found that stage-based measures of flashiness and median duration were most similar to the flow-based metrics, whereas low flow metrics had the least similarity. In an urban versus non-urban catchment comparison, they found that hydrologic variability (flashiness) was positively correlated with urban intensity index, percent developed land, population density, road density, and configuration of developed land patches and inversely correlated with forest cover. The results suggested that fragmentation (i.e., degree of scatter across basin) of developed land was negatively correlated with overall flow variability and flashiness (McMahon et al., 2003). These results are similar to the findings of Alberti et al. (2003) and provide a possible mechanistic explanation (flow variability and flashiness) for how land use patterns affect benthic communities.

Konrad and Booth (2002) studied the effect of length of record on error in hydrologic trend analyses using hydrologic metrics, derived from annual streamflow records, to study the hydrologic trends associated with urbanization. They found that trend tests on hydrologic metrics using 10 years of record or less are likely to identify trends not associated with land-use changes, and may fail to identify trends produced by urban development that occur incrementally over periods of more than a decade.

A number of researchers have advocated use of hydrologic metrics that quantify altered streamflow regimes to form mechanistic links between urbanization, hydrology, hydraulics, and biota (Scoggins, 2000; Kennen & Ayers, 2002; Eisele et al., 2003; Kirby, 2003; Booth et al., 2004; Cassin et al., 2005). Scoggins (2000) analyzed six years of

Rapid Bioassessment (RBA) data from urban streams in the city of Austin, Texas, in conjunction with continuous flow data from United States Geological Survey (USGS) gages at the outlets of the catchments using IHA metrics along with statistical procedures recommended by Poff and Ward (1989). He demonstrated the usefulness of these procedures to differentiate between levels of development and geohydrological conditions and evaluate their relationship with stream biota. Ecologically relevant parameters such as rise rate, number of high pulses, and number of zero flow days were found to be much higher in the highly developed stream than in the undeveloped stream. The results of his regression analysis showed a significant relationship between the macroinvertebrate metrics and the duration of high pulses, the date of the one-day maximum flow, percent of floods in a 60-day period, and the annual coefficient of variation.

Kennen & Ayers (2002) found that decrease in base flow, periodicity in peak discharge, and the flashiness of stream flow substantially affected the types and condition of invertebrate, fish, and algal communities in New Jersey streams, in part by their direct effects on the stream channel. Changes in mean annual peak flow were significantly related to all communities, however the two year peak discharge was found to be significantly related only to the fish- and algal-community structure.

Kirby (2003) was able to demonstrate linkages between hydrology, hydraulics, and stream biotic condition by characterizing specific aspects of the flow regime altered by urbanization using the IHA method to characterize flow regimes along an urban gradient in the Potomac River Basin. The in-stream hydraulics inferred from the IHA parameters were estimated using HEC-RAS. Finally, statistical relationships were established between specific hydrologic and hydraulic variables and macroinvertebrate community integrity. The results of a PCA identified altered flow predictability, low flows, and flashiness (rise and fall rates) as the most significant hydrologic parameters related to benthic macroinvertebrate community health. Analysis of the relationships between hydrologic change and modeled hydraulic factors indicated that hydraulic parameters including Froude number, measures of stream energy, channel width divided by drainage area, and cross-sectional flow area may be useful indicators of altered watershed condition if thresholds can be identified. Figure 2-1 illustrates the relative importance of the different classes of hydrologic variables to macroinvertebrate community integrity, based on the normalized number of statistically significant relationships observed in his study.

Booth et al. (2004), examined the relationship between land cover (TIA) and B-IBI in the Puget Sound Basin at three spatial scales: sub basin; riparian (200m-wide buffer on each side of stream for the full length of the network upstream of from the sampling point); and local (200m-wide buffer for one kilometer upstream from the sampling point). They also used hydrologic metrics to characterize annual stormflow and baseflow patterns for a 12-year period. They chose to calculate the fraction of the year that the daily mean exceeds the annual mean, T_{Qmean}, as an indicator of flashiness (lower values meaning flashier). The fraction of a multi-year period that the flow exceeds the 0.5-year return period storm, $T_{0.5}$, was calculated as a measure of the influence of urbanization because high flows tend to increase in frequency, but are shorter in duration, as a result of development; that is, individual high flow events occur more often with more development. Figure 2-2 shows that flashiness (represented by two hydrologic

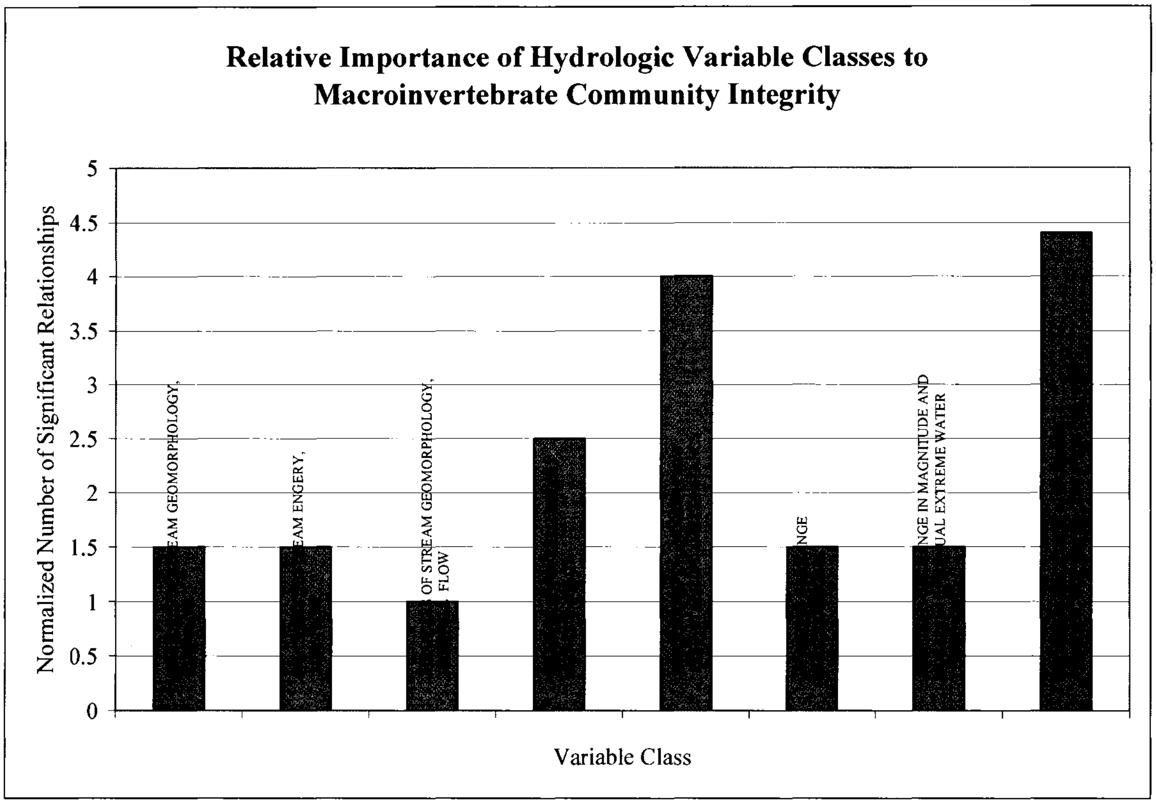


Figure 2-1. Hydrologic Variable Classes Ranked According to Their Relative Degree of Relatedness to the Measures of Biological Integrity (Kirby, 2003).

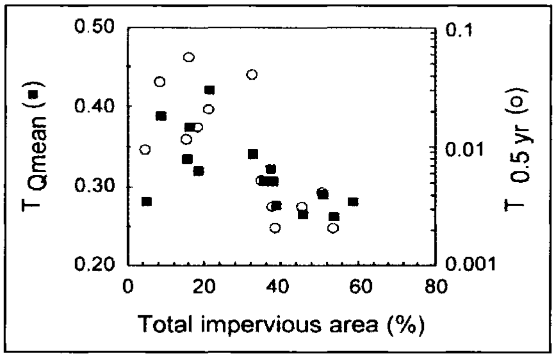


Figure 2-2. Discharge flashiness as measured by two hydrologic metrics. T_{Qmean} on the left axis; $T_{0.5}$ on the right axis; lower values are flashier (Booth et al. 2004).

metrics) increases with increased percent impervious area. The relationship between these metrics and biotic integrity for the Puget Sound area is illustrated in Figure 2-3. In this figure, *r* represents the correlation coefficient, *p* is the probability of a null

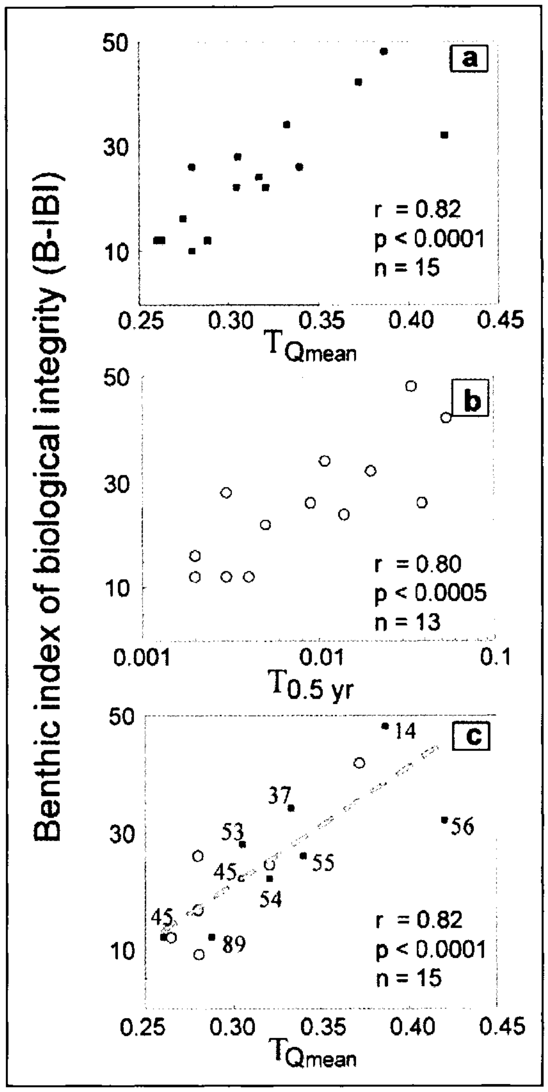


Figure 2-3. Relationship between B-IBI and (a) T_{Qmean} and (b) $T_{0.5}$. In (c) numbers indicate local urban land cover percentage – sites plotted as circles lack local land cover data (Booth et al. 2004).

hypothesis, and *n* is the number of watersheds evaluated. It can be seen that B-IBI scores are higher in less flashy watersheds.

In the King County Normative Flow Project, Cassin et al. (2005) expanded the approach from Booth et al. (2004) to develop an ecologically-based flow assessment method that could be used to evaluate the effects of past, present, and future water and land management actions on the flow regimes of streams and rivers; evaluate the effects of altered flow regimes on the physical and biological conditions in rivers and streams; and guide management actions to reduce these effects through mitigation, protection, and restoration of flow regimes that support ecosystem integrity and salmon conservation. Hydrologic metrics were computed from both stream gage data and flow data simulated by Hydrologic Simulation Program Fortran (HSPF). Hydrologic metrics were evaluated to identify those that were not different between HSPF and stream gage data. Additionally, the metrics were also evaluated to identify those with the best correlations to benthic macroinvertebrate community data. The following hydrologic metrics were identified as appropriate for computation by modeling and as predictors of biotic response:

- Low-flow threshold pulse events and interval between pulses,
- High-flow threshold pulse events and total period of the year with high pulses,
- T_{Qmean},
- Percent of time above the mean 2-year flow, and
- Timing of onset of fall flows.

As part of a WERF study Edgerly (2006) and Edgerly et. al. (2006) performed an investigation of hydrologic metrics to determine which might be most sensitive to changed land uses and stormwater runoff controls. Edgerly (2006) examined 10 hydrologic metrics proposed by various researchers to determine their sensitivity to urban runoff control strategies. The hydrologic metrics included: $T_{0.5}$, which is the percent of time that the stream flow exceeds the peak flow of the *predevelopment* 0.5-year storm; T_{Qmean}, which is the percent of time that the stream flow exceeds the mean annual flow;

high pulse duration rise rate, and fall rate metrics from Richter et al. (1996); event-mean rise and fall rate; and the ratio of event-peak-duration to total event duration. These hydrologic metrics are described in Table 2-2. It should be noted that the $T_{0.5}$ metric presented by Edgerly (2006) and Edgerly et al. (2006) is the same as the $T_{0.5}$ metric used by Cassin et al. (2005), but is different from the $T_{0.5}$ metric presented by Booth et al. (2004), which used the peak flow of the 0.5-year storm computed from the existing flow record, rather than a hypothetical predevelopment 0.5-year storm. Of the 10 hydrologic metrics examined, Edgerly (2006) found that only the T_{Qmean} was sensitive to SWMM modeling of land use change. However, the T_{Qmean} was not found to be as sensitive to stormwater runoff control strategies as the $T_{0.5}$.

Table 2-2. Summary of Hydrologic Metrics Evaluated by Edgerly (2006).

Annual Metrics	Description	Units
$T_{Q\,mean} = \left[\dfrac{\sum_{yr} t\left[\left(\dfrac{Q(t)}{Qmean}\right) > 1\right]}{1year}\right]$	The fraction of a year that the daily mean flow rate exceeds the annual mean flow rate for that year	Fraction of time
$T_{Q\,0.5\,yr} = \left[\dfrac{\sum_{yr} t\left[\left(\dfrac{Q(t)}{Q_{0.5}}yr\right) > 1\right]}{1year}\right]$	Fraction of a year that the 15-min mean flow rate exceeds a fixed pre-development half-year peak flow rate.	Fraction of time
Event-based Metrics	**Description**	**Units**
$Q_p = \max Q(t)$ from t_i to t_f	Event peak flow rate	cms
$D = t_f - t_i + 1$	Event storm flow duration	hrs
$t_p = t(Q_p) - t_i + 1$	Time-to-peak	hrs
$\dfrac{t_p}{D} = \dfrac{t(Q_p) - t_i + 1}{t_f - t_i + 1}$	Ratio of time-to-peak to storm flow event duration	ratio
$R_N = \left(\dfrac{Q_p - Q_b}{t_p - t_i + 1}\right)$	Event-mean rise rate	$\dfrac{cms}{hr}$
$F_N = \left(\dfrac{Q_b - Q_p}{t_f - t_p + 1}\right)$	Event-mean fall rate	$\dfrac{cms}{hr}$
$D_{peak} = \sum_{t=t_i}^{t_f} t\left[\left(\dfrac{Q(t)}{0.75xQ_p}\right) > 1\right]$	Duration that the flow rate exceeds 75% of the peak flow rate in an event.	hrs
$\dfrac{D_{peak}}{D}$	Ratio of event-peak-duration to total event duration	ratio

Where:

t = time-step
$t(Q_p)$ = time-step of peak flow
t_i = start time-step of event
t_f = end time-step of event

$Q(t)$ = flow rate at time=t
Q_p = peak flow rate of the event
Q_b = base flow rate
$Q_{0.5yr}$ = peak flow with 6-month return period
Q_{mean} = annual mean flow for that year

3.0 GUIDELINES FOR APPLICATION OF THE PROTOCOL

This chapter provides general guidance for application of the protocol to set stormwater management and design criteria for urban runoff control to maintain the desired level of biotic health in receiving streams in terms of the macroinvertebrate metrics. Each Step in the protocol is addressed in Sections 3.3 through 3.10 of this chapter. Section 3.8.2 provides applications of establishing relationships between land use practices, runoff control strategies, and hydrologic and geomorphic metrics. During the development of the protocol, individual hydrologic and geomorphic components were preliminarily tested using watersheds in Fort Collins, Colorado and Lenexa, Kansas. No preliminary testing was conducted on establishing relationships between biotic data and hydrologic and geomorphic metrics because data were not available; the basis for this approach is from applications in the literature (Scoggins, 2000; Kennen & Ayers, 2002; Kirby, 2003; Booth et al., 2004; Cassin et al., 2005) and the testing of the protocol in the North Carolina Piedmont. Sections 3.7 and 3.8 include examples of establishing linkages between hydrologic and geomorphic metrics and biologic data, and evaluating the impact of runoff control strategies in the North Carolina Piedmont.

3.1 Introduction to the Protocol

This protocol is built upon literature relating biologic metrics such as the benthic index of biotic integrity (B-IBI) or Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness to hydrologic metrics that describe the flow regime of urban streams (Scoggins, 2000; Kennen & Ayers, 2002; Kirby, 2003; Booth et al., 2004; Cassin et al., 2005). Additional research by Edgerly et al. (2006) showed that hydrologic metrics can be computed using the SWMM model and certain hydrologic metrics are sensitive to alternative runoff control scenarios. Building on these findings, the protocol illustrated in Figure 3-1 was developed. The black boxes comprise the identification of goals for resource protection, relevant parameters, sites used for biologic data collection and ongoing monitoring. The white boxes in Figure 3-1 are data analysis activities

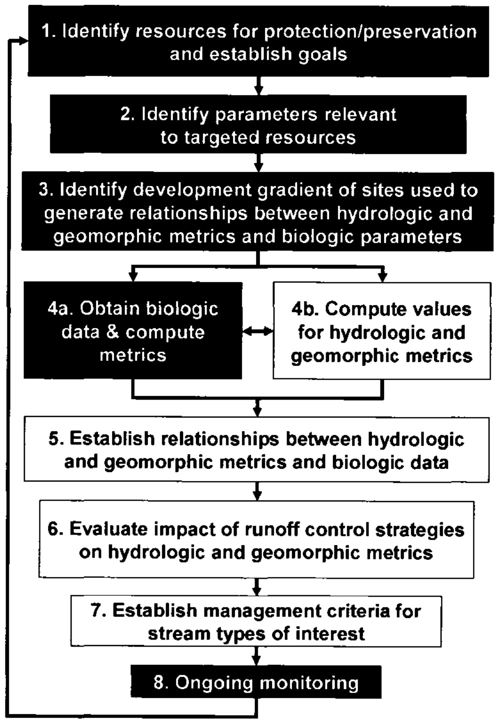


Figure 3-1. Overview of the Protocol.

comprising hydrologic, geomorphic and biologic analyses that define the baseline for stormwater management planning, and form a basis for evaluating the relative biotic impact of alternative stormwater management plans (land use patterns and/or runoff control strategies) for developing watersheds. Details of these analytical tasks are contained in Sections 3.6.2 - 3.9. It should be noted that the focus of the protocol is on the relationship of stormwater management practices in urbanizing watersheds to biological health in the receiving streams, as represented by measures of benthic macroinvertebrate communities. Water quality is not explicitly included in the protocol; however, it is included implicitly in the protocol, because the protocol requires stormwater treatment practices be employed as part of the hydrologic analysis. The eight steps of the protocol are summarized below.

- Step 1 involves setting protection goals for the target streams, discussed in greater detail in a following section, which may be performed in the context of existing state or tribal water quality standards.
- Step 2 is the identification of the biological, chemical, hydrologic and geomorphic parameters to be measured that are reflections of the goals set in Step 1.
- Step 3 involves the identification of study sites representing the full range of development conditions (from undeveloped to urban) in support of establishing stressor-response gradients for the study area.
- Step 4a involves the collection of biologic data for each study site.

- Step 4b requires continuous flow records for the computation of hydrologic and geomorphic metrics. The continuous flow records can be obtained from long-term monitoring or through hydrologic and hydraulic modeling.
- Steps 5 and 6 utilize the data collected in Steps 1 through 4 of the study protocol to develop linkages among hydrologic, geomorphic, and biologic metrics and the land use / level of development in study watersheds.
- Step 7 uses these linkages developed in Steps 5 and 6 to develop stormwater management criteria for the protection of stream resources.
- Step 8 provides guidance for ongoing monitoring to ensure that established management criteria are sufficiently protective of the target water resources. If monitoring studies indicate degradation in stream conditions relative to the acceptable management goals, an adaptive management approach is required.

3.2 Limitations of the Protocol

It is important to understand that the protocol presented herein is not the last word in evaluating the impacts of urban runoff on the biotic integrity of streams, but rather it is a tool or methodology that can be used to help stormwater managers and planners to better understand the potential impacts of implementing alternative structural runoff control strategies on the hydrologic, geomorphic and ecologic impacts in streams in urbanizing watersheds. It does not predict or simulate the biotic health of streams, but rather provides an estimate for the relative degradation in the biotic integrity of streams that might be expected if certain runoff control strategies are implemented. The strength and the weakness of this protocol is its implicit assumption that the effect of urban runoff structural controls on in-stream biotic integrity is entirely hydrologic. Water quality is not considered, except as it might be affected by pollutant removal that occurs in BMPs used to control the runoff rate from small storms. Geomorphic impacts are related to hydrologic change, but are not tied to sediment supply to the watershed.

The strength of the protocol is that it focuses on minimizing the change in the hydrologic regime of the watershed runoff in the urbanized state from that of the pre-development state. Moreover, the full hydrologic spectrum of runoff is considered, from the very small storms that transport most of the pollutant load off the watershed (WEF and ASCE, 1998) to the 100-year storm, by considering how both the peak-flow frequency curve and the flow-duration curve are modified by urbanization and how different runoff control strategies change those two curves compared to the pre-development curves. Previous studies (Nehrke and Roesner, 2003; Rohrer et al., 2005; Wulliman and Urbonas, 2005; Pomeroy and Roesner, in press; Pomeroy et al., *in press* a) have shown that using the full spectrum hydrologic control approach it is possible to develop runoff control strategies in urbanizing watersheds that reproduce the pre-development peak flow frequency curve, and most of the pre-development flow-duration curve, using state of practice peak shaving (detention) controls and a volumetric water quality control. This "full spectrum control" also reduces the erosive work that occurs in the stream channels of urbanized watersheds. Intuitively, if the hydrologic regime of the runoff from an urbanized area is controlled so that it reproduces the pre-development runoff regime of the watershed, then stream degradation can be reduced.

There are some issues that are not addressed by the protocol that a user should be aware of, these are:

- The protocol deals only with post-urbanization hydrologic conditions, it does not address the effects of changes in sediment load from a watershed. It does not address issues related to increased sediment loads that might occur during development if the proper erosion controls are not applied.
- The protocol does not account for the fact that the watershed sediment load is usually less after urbanization, than prior to urbanization due to changes in land use and/or the sediment trapping abilities of stormwater best management practices.
- It has been shown that macroinvertebrate response is strongly related to substrate (Waters, 1995; Berry et al., 2003) such as changes in sediment size due to urbanization; but since changes in the composition of stream substrate are related to the hydrology of the stream, it may be that the use of hydrologic stream metrics is still a more suitable link between urbanization and the biotic integrity of streams.
- Water quality impacts on benthic macroinvertebrates are not addressed in this protocol. An implicit assumption is that since volumetric water quality controls are used as part of the hydrologic control strategy, and that if properly designed water quality controls are used, the pollutants of concern will be removed.

The protocol is a step forward in developing tools that municipalities can use to determine the effect of their runoff control practices on the biotic health of the streams in urbanizing watersheds. There is a large body of literature on the effects of urbanization on habitat and the biotic integrity of receiving streams, but there are limited tools that stormwater managers can use to help them determine the best stormwater management practices to minimize the effects of urbanization on the receiving streams. This protocol attempts to fill that knowledge gap. As additional knowledge is gained on the cause-effect relationships of flow, geomorphology, water quality and biotic integrity in urban streams, these relationships can be incorporated into the model, improving its predictive capability and usefulness as a planning tool.

3.3 Step 1 – Identify Resources for Protection/Preservation and Establish Goals

Urbanization and the hydrologic changes associated with increased impervious (paved) surfaces have degraded aquatic life to the extent that this pattern has been dubbed the "urban stream syndrome" (Walsh et al., 2005; Meyer et al., 2005). Monitoring to identify which factors are limiting to aquatic life is crucial to protection and restoration efforts. The selection of appropriate biotic goals (Step 1 of the protocol, Figure 3-2) and adequate monitoring capabilities to accurately measure parameters that reflect these goals are critical to the entire process for protection of urbanizing waters.

The selection of biotic goals may be guided by existing state or tribe-assigned aquatic life uses for a particular water resource. The basic regulatory impetus driving the protection of streams and rivers across the United States is the Federal Water Pollution Control Act or "Clean Water Act" of 1972 and amendments (33 U.S.C. §1251 et seq.; 40 C.F.R. parts 104-149), along with associated state water quality laws and rules that further define protection provided by each state. In addition, other laws such as the

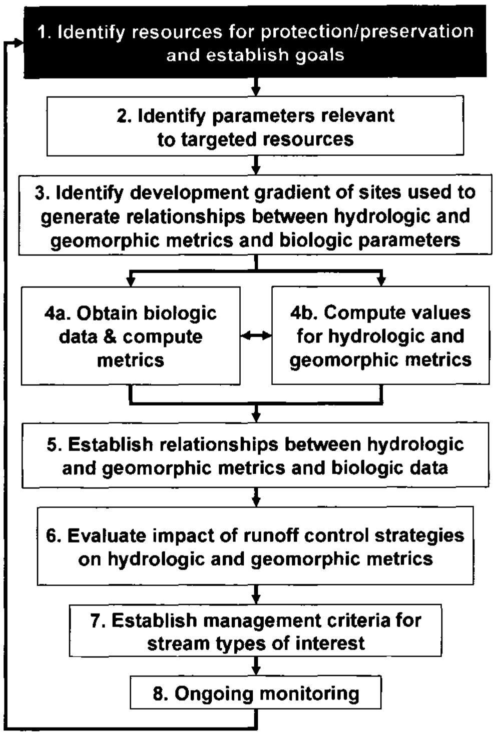


Figure 3-2. Identify Resources for Protection /Preservation and Establish Goals.

federal Endangered Species Act or "ESA" (PL 93-205; 16 U.S.C. 1531 et seq.) and similar federal regulations and state laws protecting rare and endangered species can influence levels of protection provided to aquatic life in streams and rivers.

Step 2 of the protocol calls for specific biological measures (e.g., the diversity or abundance of various elements of the fish or aquatic macroinvertebrate communities) to be identified for inclusion in monitoring efforts to determine attainment or non-attainment of biotic goals. Beyond the link to specific management goals, exploration of metrics or indices that are especially sensitive to urbanization stressors can be used to discriminate among the myriad of potentially limiting factors (Barbour et al., *in press* 2006). Further details on the identification of goals can be found in Pomeroy et al. (*in press* b).

3.4 Step 2 - Identify Parameters Relevant to Targeted Resources

Step 2 of the protocol involves the identification of study parameters to be measured. These parameters may include measures of biological community condition, hydrologic and geomorphic parameters, chemical water quality analytes, and physical stream habitat measures. The selection of study parameters must correspond with the protection/preservation goals identified in Step 1. States and tribes might have existing study measures prescribed as part of their efforts to comply with the Clean Water Act requirements. Pursuant to Steps 2 and 4a of the protocol (Figure 3-3), biological parameters and sampling locations should be selected for and collected from a given study area based on the management goals that the study is intended to support. Both selecting goals for urban waters and deciding which streams to sample and include or censor from further study requires a good conceptual model for the range of biological

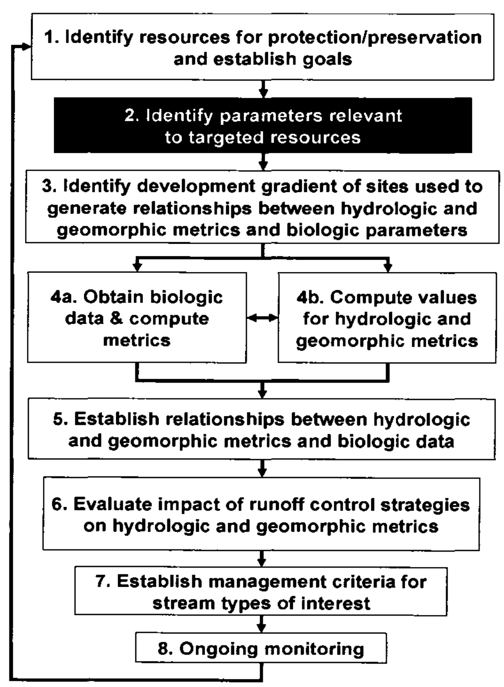


Figure 3-3. Identify Parameters Relevant to Targeted Resources.

conditions that can be expected in a region. Optimally, study sites should be selected that not only exhibit the full range of development / land use in the study area but also are representative of a biological condition gradient for aquatic life (U.S. EPA, 2005; Barbour and Yoder, draft 2005) in that region. Identification of study parameters should be predicated on established data quality objectives that are tied to the intended use of the study and associated results. Pomeroy et al. (*in press* b) provides further details on the recommended process for determining a minimum set of biological parameters to include in a study of the biological condition of streams in developed and undeveloped watersheds.

Concerning hydrologic parameters, Table 3-1 summarizes a number of hydrologic metrics determined to be most useful in predicting biological condition in six different studies from five different regions. Climatic region and stream type should be taken into consideration when determining which hydrologic metrics to evaluate. Olden and Poff (2003) presented a list of recommended indices that best represent the biologically critical portions of the flow regime. These results are summarized in Table A-1 of the Appendix. Six different stream types are shown that cover a range of regional flow regime types, including: harsh intermittent, intermittent flashy, snowmelt, snow and rain, superstable or stable groundwater, and perennial flashy. Tools for the identification of stream type are provided in Poff (1996). With the selection of appropriate metrics, this protocol can be used for snowmelt or arid streams; however, it is not applicable for watersheds with transbasin diversions, as is discussed in Section 3.5. It should be noted that the metrics presented in Table A-1 have not been evaluated against biologic data to determine their effectiveness in predicting biotic response, they are simply provided as a

Table 3-1. Comparison of Hydrologic Metrics Found to be Most Useful for Predicting Biotic Integrity.

Study	Location	Measure of Biotic Integrity	Most Useful Hydrologic Metric for Predicting Biotic Integrity	Definition	Flow Regime Characteristic Represented by Metric
Clausen and Biggs (1997)	New Zealand	Periphyton	Q_{mean} or Q_{50}	Mean Flow or Median Flow, flow exceeded 50% of the time.	Absolute Size of River
		Invertebrates	*FRE3*	Frequency of flows greater than three times the median flow.	Flow Variability and Flood Frequency
Scoggins (2000)	Austin, TX	Invertebrates	% of floods in a 60-day period	Common 60-day period for multi-year flow record that contains the largest percentage of floods (Poff and Ward 1989).	Flood Predictability
			Mean Annual CV	The average over all years of the mean flow divided by the standard deviation times 100 (Poff and Ward 1989).	Overall Flow Variability
			High Pulse Duration	Mean duration of high flood pulses, where high flood pulses are defined as the 75th percentile (Richter et al. 1996).	Flow Variability and Flashiness
			Date of High Pulse	The mean Julian date of the 1-day annual maximum flow over all years (Richter et al. 1996).	Timing/ Predictability
Kennen & Ayers (2002)	New Jersey	Invertebrates Fish Algae	*RT2575*	Ratio of discharge exceeded for the indicated percentage of time (Reed et al. 1997)	Flashiness
			2YRPK	2-year peak flow (Reed et al. 1997)	Flood Frequency
			10YRLOW	7-day, 10-year low flow (Reed et al. 1997)	Low Flows
Kirby (2003)	Virginia	Invertebrates	Rise/Fall Rates of Hydrograph	Means of all positive/negative differences between consecutive daily values (Richter et al. 1996).	Flashiness
			High Pulse Count	Number of high pulses within each year (Richter et al. 1996).	Flashiness
			Date of High Pulse	The mean Julian date of the 1-day annual maximum flow over all years (Richter et al. 1996).	Timing/Predictability
Booth et al. (2004)	Puget Sound, WA	Invertebrates	T_{Qmean}	The average annual fraction of a year that the daily mean flow exceeded the annual mean flow of the given year, which yields lower fractions for "flashy"streams and higher fractions for gradually varying flow regimes (Konrad and Booth 2002).	Flow Variability and Flashiness
			$T_{0.5}$	The fraction of time that a stream channel is exposed to flows whose magnitude exceeds the half-year flood.	Flow Variability and Flashiness

Study	Location	Measure of Biotic Integrity	Most Useful Hydrologic Metric for Predicting Biotic Integrity	Definition	Flow Regime Characteristic Represented by Metric
Cassin et al. (2005)	King County, WA	Invertebrates	Low Pulse Duration	Number of days between low pulse events, or the average number of days per occurrence that the hydrograph was below the low-flow threshold, for each calendar-year	Duration
			Low Pulse Range	The difference, in number of days, between the last and first times that the daily-time-step hydrograph crossed the low-flow threshold, for each calendar-year	Duration
			High Pulse Duration	Average number of days per occurrence that the hydrograph was above the high-flow threshold (200% of the average flow-rate under forested conditions over full period of record) for each water-year	Duration
			High Pulse Range	The difference in number of days between the last and first times that the daily-time-step hydrograph crossed the high-flow threshold (200% of the average flow-rate under forested conditions over full period of record), for each water-year	Duration
			T_{Qmean}	The percentage of time (or days per year) in a given water-year that the daily-time-step hydrograph was above the year's average forested flow-rate (Booth and Konrad 2002)	Flow Variability and Flashiness
			%of time above 2-year mean	The percentage of time that the daily or 15-minute time step was above the 2-year mean forested flow-rate	Flashiness

starting point to determine which metrics may be most effective in predicting biotic response. Collaboration with ecologists is required to identify metrics that have the greatest ecological relevance, as discussed in Sections 3.3 and 3.4.

These hydrologic metrics may be used as a point of departure in developing linkages among land use, hydrologic, geomorphic, and biologic conditions. The hydrologic metrics are computed from a continuous flow record. If a continuous monitoring flow record of sufficient length is not available, the protocol requires that hydrologic and hydraulic models be developed for study watersheds to simulate a long-term continuous flow record for each. See Section 3.6.2 for a more detailed discussion of hydrologic sampling, model development, and metric development.

Erosion potential analysis can be used as a geomorphic metric to evaluate the impacts of urbanization and stormwater management practices. In addition, other aspects of stream substrates may be measured or modeled to evaluate the potential impacts of urban development. The substrate metrics could range from pebble counts to visual metrics such as the Ohio EPA Qualitative Habitat Evaluation Index (QHEI) embeddedness metric. (Rankin, 1989). Estimates of critical shear stress values, based on particle sizes or measured via in-situ jet-testing (Hanson and Cook, 2004) can be compared to calculated shear stress values using stream-specific erosion potential indices (Bledsoe, 2002). These indices can be utilized for the development of linkages among land use and hydrologic, geomorphic and biologic parameters. The information might also be utilized in the development of management criteria (Figure 3-2, Step 7). Erosion potential indices could also be calculated for watersheds targeted for protection in followup surveillance monitoring (Figure 3-2, Step 8) to evaluate the effectiveness of management efforts to protect against stream channel erosion.

3.5 Step 3 – Identify Development Gradient of Sites Used to Generate Relationships between Hydrologic and Geomorphic Metrics and Biological/Habitat Parameters

During Step 3 of the protocol (Figure 3-4) sites should be selected to represent a gradient of urbanization. The selected watersheds should represent a range of undeveloped, partially developed, and fully developed conditions as well as differences in density of development such as low, medium or high intensity land use. If continuous flow monitoring data are used in the protocol to compute hydrologic and geomorphic metrics, watersheds should be selected that have not had significant development or land

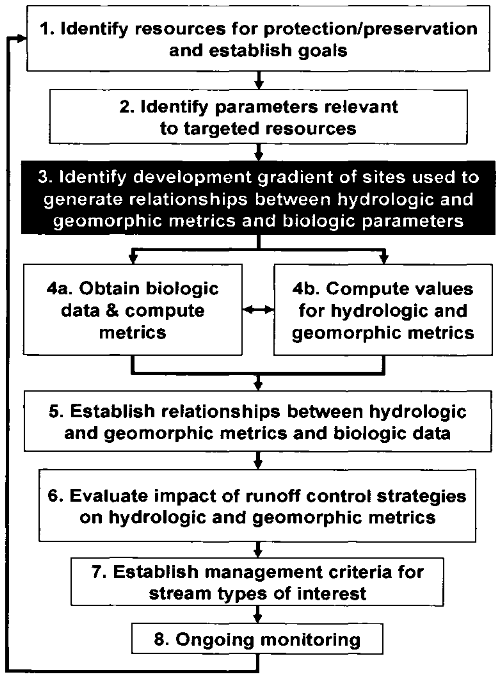


Figure 3-4. Identify Development Gradient of Sites.

use change during the monitoring period of record. If significant land use changes have occurred, a rainfall-runoff model can be used to develop a flow record that is based on existing land use in the watershed, rather than changes in land use over time.

For a regional study, at least 6 to 10 watersheds should be identified that meet the following criteria:

- Similar climatology, or situated in similar climatologic areas.
- Homogeneous land use within each watershed – The intent of this criterion is to maximize the potential for a study to identify impacts of different land use or development types. Urban and suburban watersheds commonly have a mix of low, medium or high density residential land uses intermixed with commercial land use, as well as urban open space such as parks, schoolyards or cemeteries. Identification of homogenous land use in the watershed in not intended to select watersheds with the same land use throughout the watershed, rather, watersheds should be avoided that contain clustering of land use which could include high, medium or low intensity residential development in the headwaters of the watershed and forest or agricultural land near the outlet of the watershed such as is shown in Figure 3-5a, or a split of urban land use along one tributary and no urban development along the other, such is shown in Figure 3-5b. Figure 3-5c and Figure 3-5d show watersheds with a spatially distributed mix of land uses.
- Similar stream geomorphic characteristics – To the extent possible, streams in selected watersheds should exhibit similar geomorphic characteristics, regardless of the level of development in the watershed. Such characteristics

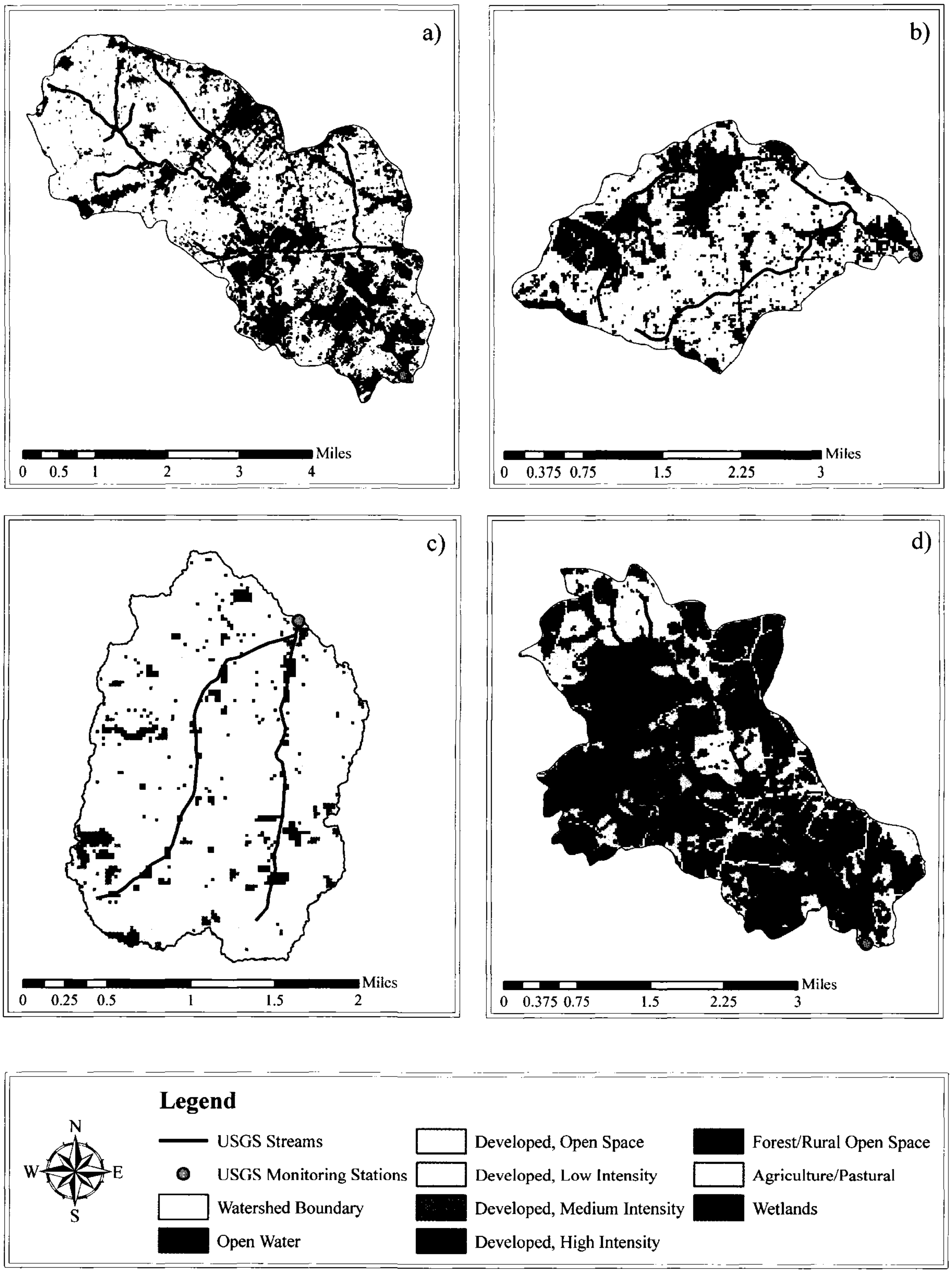


Figure 3-5. Examples of Land Use Distribution.

should include substrate size, similar geomorphic channel units, and local geologic setting.

- Watershed size and stream order – Study watersheds generally should have an area no greater than approximately 5 - 10 square miles, with no larger than 3rd order streams. The rationale for this criterion is that larger streams tend to be somewhat buffered against the hydrologic impacts of development due to response of a more complex watershed system. Additionally, if development and stormwater management practices are not implemented uniformly throughout a watershed their impacts on hydrologic and geomorphic metrics may be difficult to discern. Streams with drainage areas of less than 10 square miles also tend to be wadeable streams that are accessible under normal flow conditions without the need to use a boat for field sampling. Wading methods may be used to sample streams with drainage areas as large as 150 - 200 square miles; however, sampling sites with such large upstream drainage areas require consideration of drainage area impacts related to changes in biotic assemblages.
- Presence / absence of stormwater management practices – Watersheds should be selected both with and without established stormwater management practices (BMPs), particularly BMPs that improve water quality. The intent of this criterion is to enable studies to clarify the potential for attenuating the impacts of development with stormwater management approaches.
- Avoidance of non-stormwater related sources of potential impairment – Watersheds with significant point source discharges, combined sewer

overflows (CSOs), dams, transbasin diversions and other significant river engineering activities should be avoided.

- Flow gage availability – Watersheds with flow gages should be preferred over watersheds without, with gages preferably in close proximity to biological sampling stations. The intent of this criterion is to support calibration of hydrologic models to simulate flows at biological sampling stations.
- Rain gage availability – Watersheds with rain gages should be preferred over watersheds without gages, in support of hydrologic modeling potential. If flow data are not available, simulation of a continuous flow record requires data from continuously recording rain gages with at least 5 years of hourly or 15-minute (preferred) data.
- Elimination of acutely impaired sites - Study sites should be screened to avoid the inclusion of acutely impaired sites that exhibit water quality that is sufficiently poor to potentially obscure the effect of other development-related stressors. Details on water quality parameters for screening are given in Pomeroy et al. (*in press* b).

Sites should be selected that represent a gradient of urbanization from undeveloped to full build-out. The extent of urbanization can be determined using many parameters such as gross impervious area (GIA), directly connected impervious area (DCIA), population, housing unit or road density, percent of urban land cover or a combination of these factors. The Urban Intensity Index (UII) described by McMahon and Cuffney (2000) combines these factors to compare the extent of urbanization of watersheds in urban gradient studies conducted by the United States Geological Survey

(USGS). Factors describing the extent of urbanization in watersheds can be used as tools for the preliminary identification of candidate watersheds for monitoring. Further screening of these candidate watersheds is required with respect to geomorphic and land use homogeneity of the watersheds, sampling sites that are typical of streams in the area, with suitable locations for flow measurement available close to the biologic/water quality sampling sites.

If it is not economically or physically feasible to study a gradient of urbanization, paired watersheds or a single watershed may be used to apply the protocol. Hydrologic and geomorphic metrics can be developed for a single, undeveloped watershed, and development scenarios could be tested using hydrologic and hydraulic models to identify management practices that closely reproduce the desired hydrologic flow regime. Similarly, a pairing of undeveloped and developed watersheds can be used to identify targets for rehabilitation of a developed and degraded watershed. It is important to note, however, that there is a level of uncertainty associated with the computed hydrologic and geomorphic metric values, and that use of one data point as a representative target does not allow for a range of target values to be identified.

3.6 Step 4 - Development of Data to Support Linkages

3.6.1 Step 4a – Obtain Biological Data for Each Site and Compute Biologic Metrics

Pursuant to Steps 2 and 4b of the protocol (Figure 3-6), biological parameters and sampling locations should be selected for and collected from a given study area based on the management goals that the study is intended to support. The biologic metrics selected will be regional or state dependent. Both selecting goals for urban waters and deciding

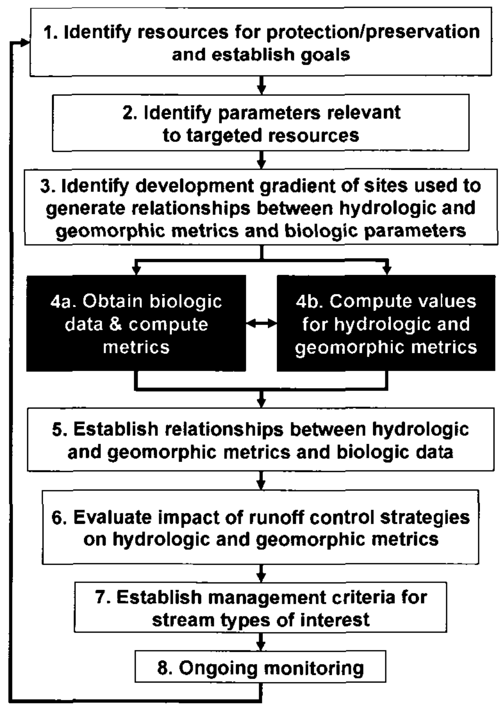


Figure 3-6. Development of Data to Support Linkages.

which streams or sample locations to exclude or censor from further study requires a good conceptual model for the range of biological conditions that can be expected in a region. As noted under Step 3, study sites should be selected that not only exhibit the full range of development / land use in the study area but that are also representative of a biological condition gradient for aquatic life (U.S. EPA, 2005; Barbour and Yoder, draft 2005) in that region.

For example, Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness data gathered by the USGS under the National Water Quality Assessment (NAWQA) program in the Effects of Urbanization on Stream Ecosystems (EUSE) study in the North Carolina Piedmont showed that of 30 sites monitored, the highest rating observed was good-fair, and the remainder were fair or poor. In this study, watersheds along a gradient of urbanization were examined (see Section 4.0), but even undeveloped watersheds did not generate EPT richness scores above a good-fair rating, as shown in Figure 3-7 . This indicates that the range of biological conditions expected may not be higher than good-fair, and that this may be the best rating that is possible to achieve, and should be incorporated into the goals (see Section 3.3).

If it is unclear which biological index is the best to use, it is suggested that the data collection program include at a minimum the data required to compute metrics for benthic macroinvertebrates, fish, or algae that are indicative of water quality in the particular region of interest. Macroinvertebrate metrics such as EPT richness, EPT percent richness, or a benthic index of biotic integrity (B-IBI) have been found to be useful in various regions of the United States, but may not be appropriate in all regions; therefore regional calibration of appropriate biologic metrics is necessary. The biologic data must be synthesized into indices that will be used as measures of the biotic health of streams draining each watershed. Further details on the recommended process for

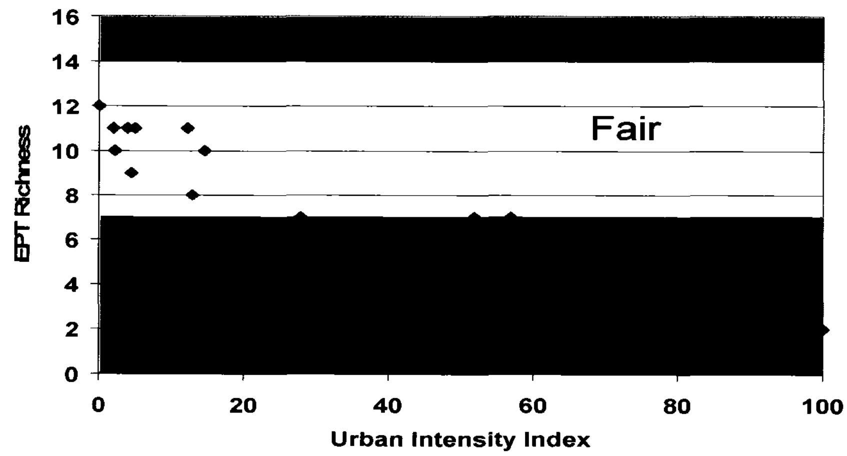


Figure 3-7. Range of EPT Richness Scores in North Carolina EUSE Study.

determining biological parameters for support of the protocol are given in Pomeroy et al. (*in press* b).

3.6.2 Step 4b - Compute Values for Hydrologic and Geomorphic Metrics for Each Site

Step 4b involves working with a continuous flow record to compute hydrologic and geomorphic metrics. If a continuous flow monitoring record of at least five years of hourly- or 15-minute data is not available for each sampling site, the data set can be extrapolated using computer simulation models to represent the complete hydrologic and geomorphic character of the watersheds being studied. If flow monitoring is less than five years, hydrologic metrics computed from that short hydrologic record will not reflect the true value of the hydrologic metric because it will not reflect the full range of storms that occur in the watershed. However, the biologic data are likely reflect the hydrology of the last three to five years, therefore it is recommended that the hydrologic metric data be computed using a five year record, unless that record includes a very large storm, a very wet period, or a severe drought. In addition, a longer period of record is recommended for computation of geomorphic impacts, so that the effects of large (> 5-year return period) storms can be used to evaluate changes in the potential for erosion on the stream channel. Though it is generally believed that the channel forming discharge occurs between a one to two-year return period, the effects of larger events are not inconsequential and should be considered when evaluating geomorphic metric values.

3.6.2.1 Development of Continuous Simulation Models for Computing Hydrologic and Geomorphic Metrics

There are a variety of hydrologic and hydraulic mathematical models that can be used for continuous simulation of stream flow in watersheds. Two of the most popular are Hydrologic Simulation Program Fortran (HSPF), and EPA Stormwater Management Model (SWMM). Other models can be applied, but they must meet the following criteria.

- Be able to process a continuous 20- to 50-year record of 15-minute rainfall data and translate it into runoff accounting for land use practices within the watershed.
- Simulate infiltration processes that may vary across the watershed.
- Simulate stream interaction with groundwater and interflow.
- Route flows dynamically through the stream channel drainage system.
- Simulate the effect of runoff controls, such as BMPs and detention storage, within subcatchments, on the runoff hydrographs from the subcatchments.
- Be able to provide a time series of channel hydraulic properties (at 15-minute intervals), including stage and discharge, during runoff and dry weather periods.

SWMM5 is well suited to these criteria; therefore it was used in the application of the protocol to the Raleigh-Durham, NC area. Use of SWMM5 for application of the protocol is explained in the remainder of this section. For guidance in using other computer models, please refer to the appropriate guidance documentation for those models.

Figure 3-8 is a logic diagram that can be used to assist a researcher applying a hydrologic and hydraulic model to a watershed. Note that under certain conditions it may not be possible to apply the model to the desired watershed, in which case that site cannot be used to relate biologic metrics to hydrologic metrics for the study area. This situation would occur if a continuous rainfall of sufficient duration (at least 5 years) is not available, or if sufficient synoptic measurements of rainfall and stream flow to calibrate the model are not available and an uncalibrated model is not acceptable. The data required for watershed modeling in SWMM5 is illustrated in Figure 3-9. Additional guidance can be found in the SWMM5 Documentation Manual (Rossman, 2005), or in Huber and Dickinson (1998).

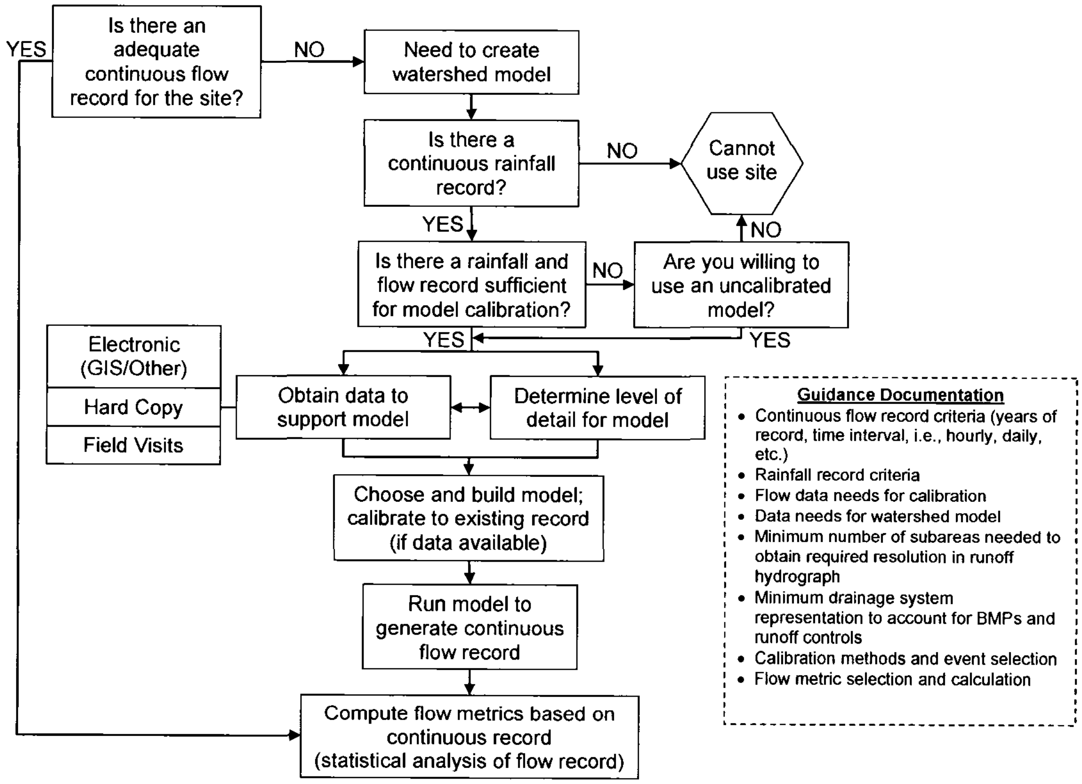
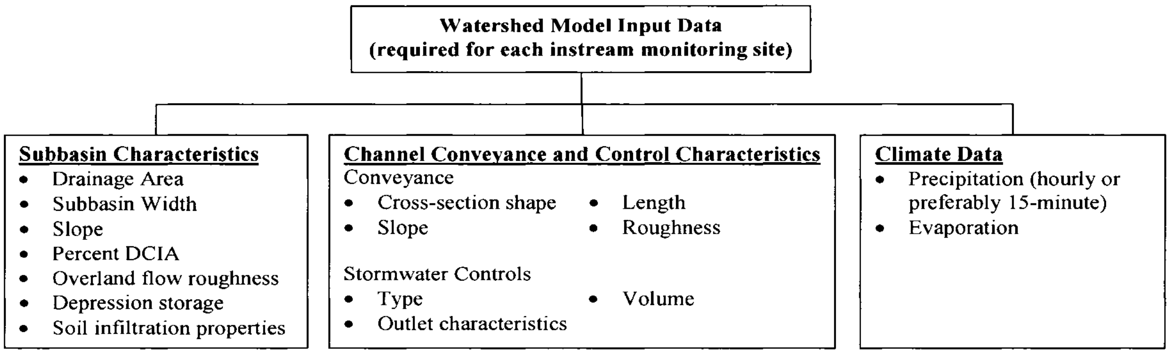


Figure 3-8. Logic Diagram for Applying a Simulation Model for Computing Flow Metrics.



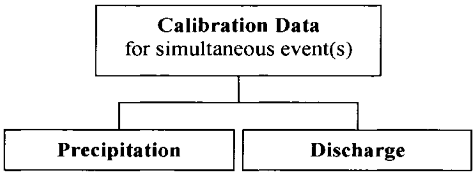


Figure 3-9. Data Requirements for SWMM5 Modeling.

SWMM and other hydrologic and hydraulic models have the ability to evaluate the hydrologic impacts of land use change and stormwater management practices. However, these models should be used with caution for the computation of hydrologic and geomorphic metrics of biologic interest. Many factors influencing historical or natural flow regimes (i.e., large woody debris, beavers, wetlands) as well as factors influencing current and future flow (i.e., influence of groundwater withdrawals on summer baseflows) may not be accurately estimated by these models (Cassin et al., 2005).

3.6.2.2 Calibration of Continuous Simulation Models

Calibration is the adjustment of watershed parameters in a model to generate a modeled behavior that best matches the monitored behavior of a watershed over a variety of runoff events. In this case, the process implies adjusting model parameters (such as infiltration rates and imperviousness levels) to obtain modeled results that match or approximate measured discharge records. Table 3-2 describes the relative sensitivity of the three portions of the flow duration curve to model parameters commonly used for calibration. These estimates were based on a simulation period of approximately 18-months.

Calibration can be done at a number of temporal scales (see Figure 3-10):

- Single event: one storm.
- Multiple events: several storms.
- Continuous simulation: long records (months to several years).

Table 3-2. Relative Sensitivity of Flow Duration Curve to Select Model Input Parameters.

Parameter	Large Flows (< 0.1 % of time)	Mid-Range Flows (0.1 – 1 % of time)	Low (> 1% of time)
Watershed Parameters			
Horton final infiltration, f_c	High	Low	None
Horton initial infiltration, f_o	Very Low	Very Low	None
Horton decay, k	High	Medium	None
Overland roughness, n	High	None	None
Percent Imperviousness	Low	Medium	Low
Aquifer Parameters			
Porosity	None	Low	Low
Wilting Point	None	High	High
Field Capacity	None	Low	Low
Hydraulic Conductivity	None	High	High
Groundwater Equation			
GW Flow Coefficient	None	Medium	Medium
GW Exponent	None	Medium	Medium
SW Flow Coefficient	None	Sensitive but produces large GW and routing errors	Sensitive but produces large GW and routing errors

Source: Davis (*in press*).

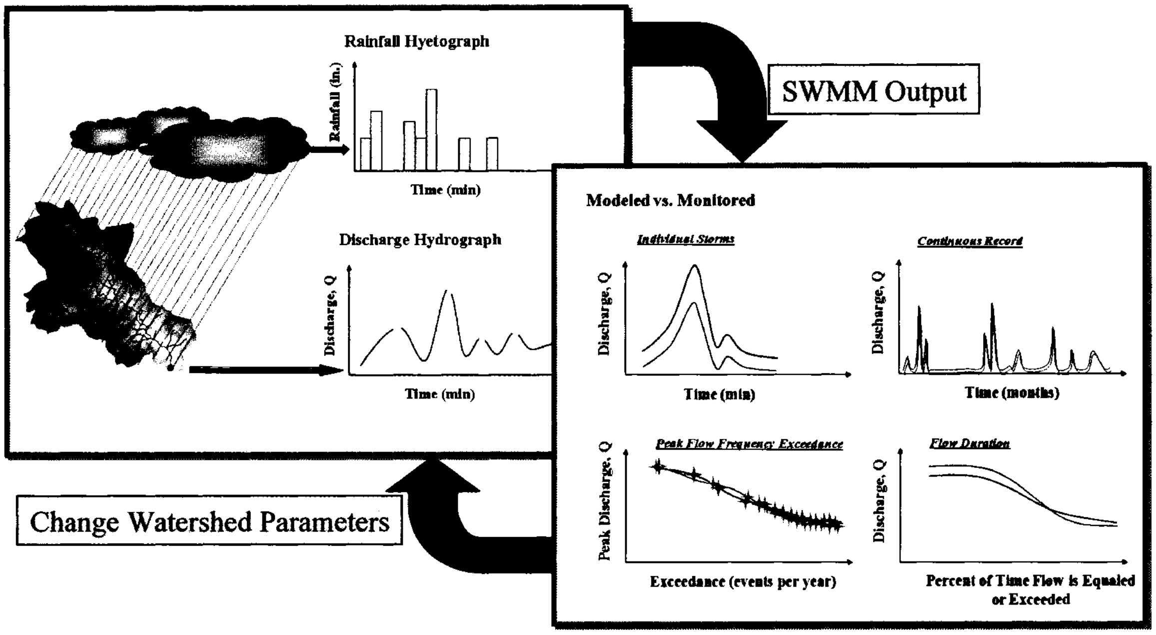


Figure 3-10. Calibration at a Variety of Temporal Scales.

Calibration to a single storm is not appropriate for continuous simulation modeling because adjustment of variables to match runoff from one event may over or under adjust variables and inhibit matching of other events. If it is desired to use individual storms for calibration, it is recommended that a minimum of three to five storms be used. One or two of the storms should be small, so that the only runoff that occurs is off the impervious areas. The other two or three storms should be large enough so that runoff also occurs from the pervious area, but not so large that flooding occurs in the watershed. Additionally, these storms should occur after at least seven days of no rainfall so that the soil is dry, and the maximum infiltration rate can be estimated.

If the rainfall data that will be applied to the model are from a gage that is more than one mile away from the centroid of the watershed, it is recommended that an initial calibration be done using five discrete storms, but that the final calibration be conducted by comparing peak-flow frequency exceedence and flow-duration curves from the model to those developed from measured data. The idea is that the rainfall on the watershed might not correspond exactly to the rainfall measured at the gage for any given storm because of the distance from the gage. Over a longer period of time the rainfall frequency distribution on the watershed being simulated is likely to be similar to the rainfall frequency distribution at the gage supplying the rainfall data for the model, so the simulated and measured peak-flow frequency and flow-duration curves should be similar.

The effect of snowmelt is also an important factor to consider for model calibration in regions where snowmelt affects streamflow. Although flow rates are typically low, they may be sustained over several days. Rainfall events superimposed upon snowmelt baseflow may produce higher runoff peaks and volumes as well as add to the melt rate of the snow. Both SWMM and HSPF have the ability to perform snowmelt computations while running continuous simulations (Huber and Dickenson, 1998; Bicknell et al., 1997).

3.6.2.3 Computation of Hydrologic Metrics

Hydrologic metric data must be developed from the flow record measured prior to and during the biologic/water quality data collection program, or from continuous computer simulation models. Under Step 2, a number of different hydrologic and geomorphic metrics were identified, which various investigators have determined relate to stream biotic health.

Edgerly (2006) and Edgerly et. al. (2006) found that that hydrologic metrics have traditionally been computed using daily mean values of flow. While this method of computation is valid for large watersheds, the rainfall-runoff response of the size watersheds recommended for this protocol (less than 10 square miles) is so rapid that trends in hydrologic metrics are not easily detected using a mean daily flow time series. To make the statistics meaningful for watersheds of this size, it is recommended that hydrologic metrics be computed from 15-minute or hourly flow data since the streams in the size watersheds recommended in Step 2 for sampling will respond to these flow changes over these short time intervals. If watersheds used in the protocol are smaller than 3 square miles, investigators should evaluate the sensitivity of the metrics computed from data at both hourly and 15-minute resolution to determine if hourly data are adequate, or if 15-minute data are required.

Computation of hydrologic metrics is best performed using a program capable of handling large data sets such as SAS™ or MATLAB™. Several programs currently exist for the computation of hydrologic metrics: the National Hydrologic Assessment Tool (NATHAT) (Henrickson, 2006), the Indicators of Hydrologic Alteration (IHA) (The Nature Conservancy, 2007), and GeoTool (Bledsoe et al., in press). NATHAT and the IHA software work only with daily streamflow data. GeoTool is able to process data at various temporal resolutions; however, the hydrologic metrics are only computed at daily increments. Unfortunately, a daily resolution is not adequate to represent changes in land use or stormwater management practices in watersheds of the size recommended in this protocol, therefore these programs are not applicable to the computation of hydrologic metrics using 15-minute or hourly time steps.

3.6.2.4 Computation of Geomorphic Metrics

Excess shear stress and erosion potential indices are geomorphic metrics that can be used to evaluate the impacts of urbanization on stream channels. Other geomorphic metrics include the size and gradation of bed substrate or descriptors of embeddedness. Shear stress duration curves can be used to help evaluate the full range of shear stress forces exerted on a stream channel, such as the one shown in Figure 3-11 for Cedar Creek in Lenexa, Kansas (metropolitan Kansas City). The shear stress duration curve is generated by calculating average boundary shear stress in the stream channel for each time step in the measured or modeled flow record. Average boundary shear stress, τ_o is calculated as:

$$\tau_o = \gamma R_h S_f \tag{3-1}$$

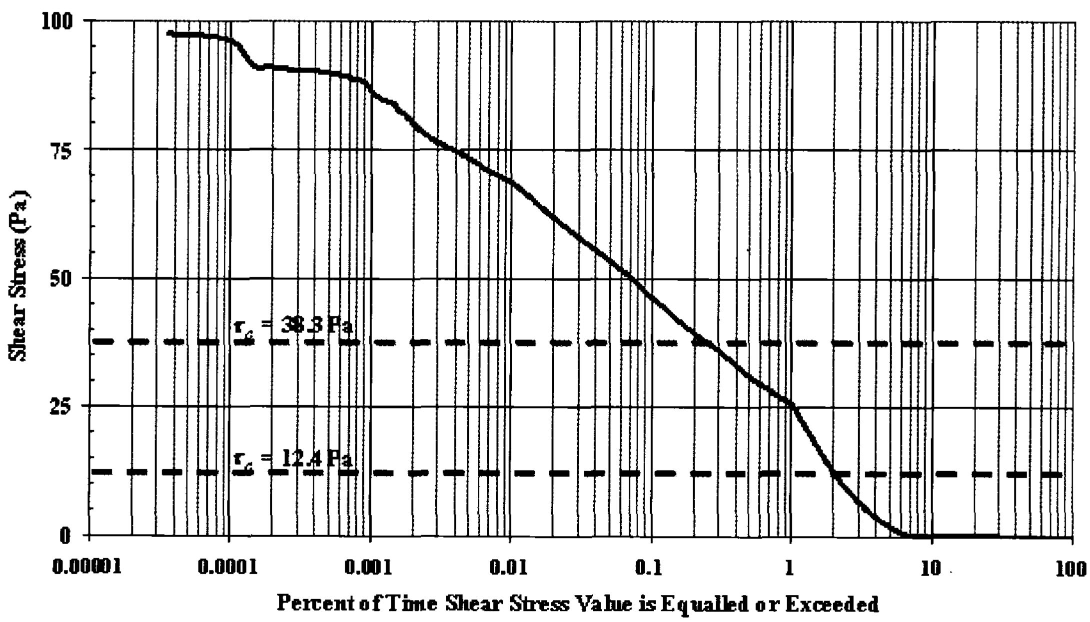


Figure 3-11. Shear Stress Duration Curve for Cedar Creek in Lenexa, Kansas.

where, γ is the unit weight of water, R_h is the hydraulic radius of channel, and S_f is the energy gradient in the channel.

The association between water and sediment discharge, particle size, and channel bed slope can be described using the relationship described by Lane (1955):

$$QS \propto Q_s\, d_s \tag{3-2}$$

where Q = water discharge, S = channel bed slope, Q_s = sediment discharge, and d_s = particle size. When urbanization occurs, the balance between sediment transport capacity and the amount of sediment delivered from a watershed is disrupted. As instream discharges increase due to urbanization of a watershed, the ability of a stream to transport sediment increases. For example, if Q represented in Equation 3-3 increases due to urbanization, and if S and d_s remain constant, Q_s will increase.

While there are numerous ways to evaluate the balance of sediment supply and delivery, channel erosion can be simplified as a function of the erodibility of the bed and bank materials and excess shear stress. Erosion rates can be estimated using an excess shear equation described by Foster et. al. (1977):

$$\varepsilon = k\left(\tau_o - \tau_c\right)^A \tag{3-3}$$

where ε is the erosion rate, k is an erodibility or detachment coefficient, τ_c is the critical shear stress, and A is an exponent, which is typically assumed to be 1.5. When performing erosion potential analysis, the value of k is factored out, and can be assigned a value of 1.0 to simplify calculations. For granular (non-cohesive) materials, τ_c can be estimated using the Shields criteria:

$$\tau_{*c} = \tau_c / (\gamma_s - \gamma) d \tag{3-4}$$

where τ_{*c} is the critical dimensionless shear stress, γ_s is the unit weight of sediment, γ is the unit weight of water, and d is the representative particle diameter.

Critical shear stress values can be used to identify the portion of a shear stress duration curve that must be matched to minimize increases in erosion potential in a stream channel, as shown in Figure 3-11. Approximate threshold values of τ_c for granular material at 20 degrees Celsius are given in Table 3-3. It should be noted that the values in Table 3-3 are based upon the classical relations for the threshold of motion of uniform sediment presented by Shields (1936), with an average τ_{*c} of 0.047. Parker (*in press*) suggests that the data presented by Shields (1936) tends to overpredict the critical dimensionless shear stress value, and that τ_{*c} is more appropriately 0.03. Given the uncertainty associated with hydrologic and hydraulic modeling and the overall variability of stream channel substrate, use of the values in Table 3-3 are adequate for computation of erosion potential until revised values of critical shear stress become widely available for geomorphic computations.

It is more difficult to estimate the erosion resistance of cohesive materials. Various studies on the erodibility of cohesive materials have shown that numerous soil properties influence erosion resistance, including antecedent moisture, clay mineralology and proportion, density, soil structure, organic content, and pore and water chemistry (Grissinger, 1982). Erodibility coefficients and critical shear stress values for cohesive sediments can be estimated by in-situ jet-testing using the methods described by Hanson and Cook (2004).

Table 3-3. Approximate Threshold Conditions for Granular Material at 20 °C.

Class Name	d_s (mm)	τ_c (Pa)
Boulder		
Very large	>2,048	1,790
Large	>1,024	895
Medium	>512	447
Small	>256	223
Cobble		
Large	>128	111
Small	>64	53
Gravel		
Very coarse	>32	26
Coarse	>16	12
Medium	>8	5.7
Fine	>4	2.71
Very Fine	>2	1.26
Sand		
Very Coarse	>1	0.47
Coarse	>0.5	0.27
Medium	>0.25	0.194
Fine	>0.125	0.145
Very Fine	>0.0625	0.110
Silt		
Coarse	>0.031	0.083
Medium	>0.016	0.065

Source: Julien (1998).

The potential for erosion in a channel can be quantitatively evaluated as the difference between calculated shear stress and critical shear stress. An instream erosion potential index can be used to estimate an increase or decrease in erosion potential by calculating a ratio of post-development to pre-development "work" on a channel (Bledsoe, 2002). This "work" on a channel can be estimated using erosion rates. The erosion potential index is calculated as a ratio of the sum of excess shear (ε, for each flow record in a continuous time series). Similar to the computation of hydrologic metrics, the flow record can come from monitoring data or from the output from a continuous simulation model. The erosion potential index uses a simple finite-difference approximation to estimate the time-integrated sediment transport capacity over the duration of the continuous flow record:

$$E = \frac{\sum_{t=0}^{T} \varepsilon_{post}}{\sum_{t=0}^{T} \varepsilon_{pre}} \tag{3-5}$$

where E is the instream erosion potential; ε is the erosion rate at time t; T is the length of the continuous flow record; and *pre* and *post* represent pre- and post-development conditions, respectively.

Erosion potential is a geomorphic metric that takes into account the effects of changes in flow regime of a stream channel, but it does not account for the effects of sediment supply or other factors likely to affect stream stability. Different types of urban streams are likely to exhibit varying degrees of instability, depending on entrenchment, relative erodibility of bed and banks, riparian condition, mode of sediment transport (bedload versus suspended load) and proximity to geomorphic thresholds (Bledsoe and Watson, 2001). Changes in land use (i.e., from agricultural to residential) combined with the sediment trapping efficiency of structural BMPs, such as extended detention ponds, may reduce bed material loads from the watershed. This reduction in bed material load, especially when combined with changes in flow regimes (as described by the erosion potential metric) can exacerbate channel instability, especially in mixed- or suspended-load channels. Additionally, a reduction in bed material load from the watershed means that reproducing the predevelopment flow regime may not maintain the stability of the stream channel, especially for those comprised of finer grained materials such as sand (Bledsoe, 2002).

3.7 Step 5 - Establish Relationships between Hydrologic and Geomorphic Metrics and Biologic Data

Relationships between the hydrologic and geomorphic metrics and the biologic parameters are established during Step 5 of the protocol (Figure 3-12). Methods for developing these relationships include regression, logistic regression, classification and regression tree (CART) analysis such as used in the WERF IIA process (Paulson et al., 2001), or using Bayesian networks (Borsuk et al., 2004).

The easiest method to develop linkages between hydrologic and geomorphic metrics and biologic parameters is by simple regression. Regression determines the form and strength of a relationship between two variables; the intent is to predict a value of a dependent variable from a given independent or predictor variable (Dytham, 1999). In this protocol, the dependent variables are the biologic metrics and the independent variables are the hydrologic and geomorphic metrics. For example, Figure 3-13 shows a regression of qualitative multihabitat EPT richness data (dependant variable) collected by the USGS in eight watersheds in the North Carolina piedmont and the T_1 hydrologic metric computed from a 20-year SWMM simulation for the same watersheds. Qualitative multihabitat samples are those collected from multiple types of habitat in a stream, as opposed to samples collected only in riffles, which provide a rich habitat environment for macroinvertebrates. The T_1 hydrologic metric is computed as the percent of time the flow is above the 1-year return interval peak discharge computed for each watershed for the period of record examined (20-years, in this example). Further details on the biologic data and computation of the T_1 hydrologic metric are given in Section 4.0.

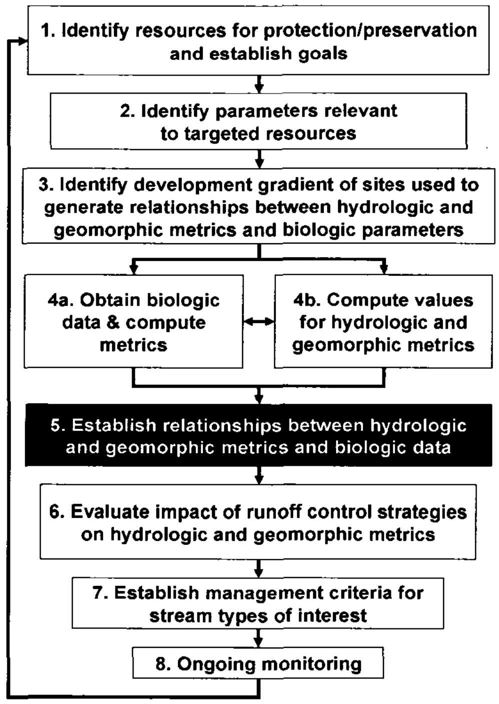


Figure 3-12. Establish Relationships Between Metrics and Data.

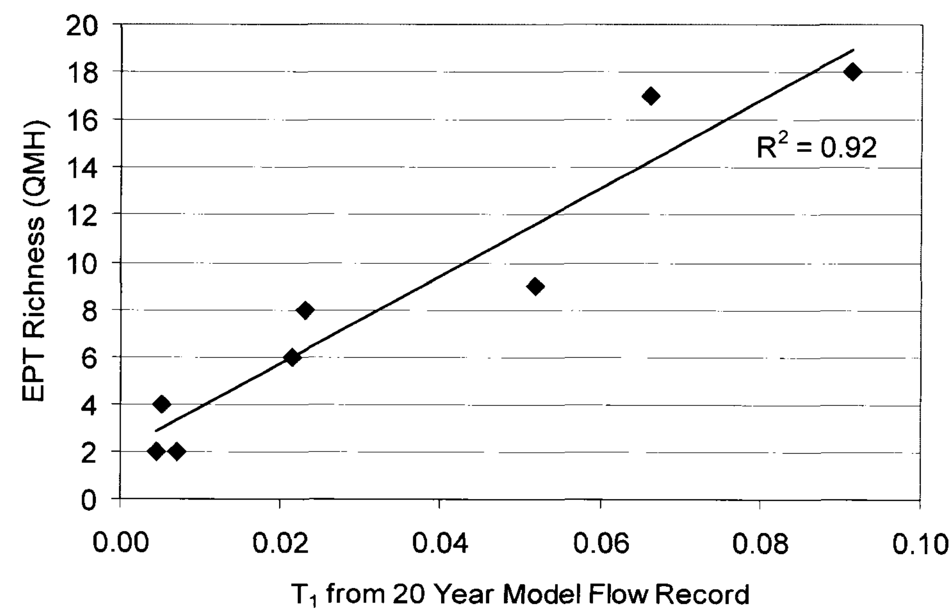


Figure 3-13. Relating Macroinvertebrate Data to the T_1 Hydrologic Metric Using Linear Regression.

Regression is the recommended methodology for this protocol. Regression is easily applied and understood, and was recommended by the reviewers of this protocol as the method most appropriate for use by municipal stormwater agencies. Additionally, regression has been proven useful in relating biologic metrics to hydrologic metrics in studies across the United States (Cassin et al., 2005; Booth et al., 2004; Scoggins, 2000).

Sample size should be taken into account when selecting the number of metrics to evaluate. Fewer sites will mean a higher level of uncertainty in the results. A general rule-of-thumb for building statistical models is to have three data points for each variable evaluated for inclusion in the model, and when using the statistical models to have 10 sites for each independent variable evaluated (Ott and Longnecker, 2000). A recommendation of 6-10 sites for this protocol was made in Section 3.3, keeping in mind the potential resource limitations of municipal stormwater management agencies. If more sites are included in the protocol analysis, additional variables can be examined. Additional sites could also be used for cross-validation or testing of the relationships developed. At a minimum, relationships between causal variables should be examined, and users should keep in mind that relationships between hydrologic and geomorphic metrics are confounded by other factors, such as water quality, and that the hydrologic and geomorphic metrics described herein may not be the sole drivers behind the biologic results.

3.8 Step 6 – Evaluate Impact of Runoff Control Strategies on Hydrologic and Geomorphic Metrics

3.8.1 General Guidance

Step 6 is the planning and analysis task (Figure 3-14). Here various land use and stormwater management scenarios are created and tested to determine their effect on hydrologic and geomorphic metrics. Examples of the stormwater management scenarios that can be evaluated include use of low-impact development practices or traditional BMPs such as extended detention ponds, swales, or constructed wetlands. The basic algorithm for this testing is illustrated in Figure 3-15. This testing is conducted using hydrologic and hydraulic models with continuous precipitation records to generate a continuous flow record. Resulting metrics computed from the continuous flow record for the various management scenarios are then compared against metric targets to determine which scenarios most closely match targeted goals. This process is part of Step 7 of the protocol which provides guidelines for establishment of management criteria.

3.8.2 Examples of Application

3.8.2.1 Impact of Runoff Control Strategies on Hydrologic and Geomorphic Metrics in a Hypothetical Development in Fort Collins, Colorado

Hydrologic and hydraulic modeling in SWMM4.4h were used to evaluate the effectiveness of typical stormwater management practices on hydrologic and geomorphic metrics for a 10 ha watershed in Fort Collins, Colorado (Rohrer, 2004; Rohrer and Roesner, 2006; Edgerly et. al., 2006). Fifty-year continuous simulations with

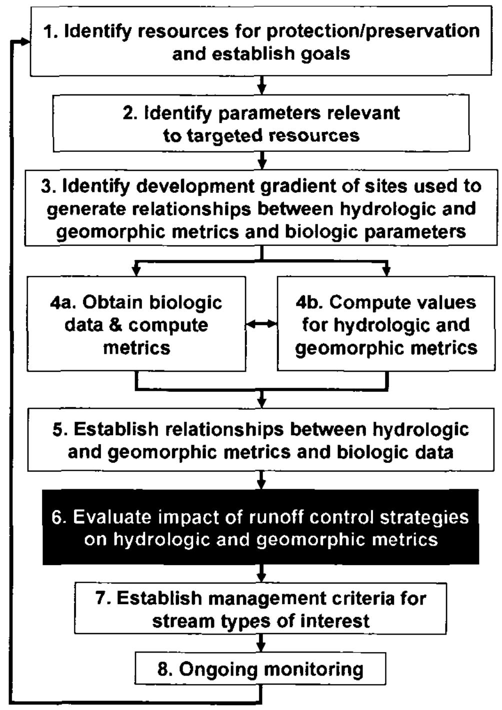


Figure 3-14. Establish Relationships Between Management Practices and Metrics.

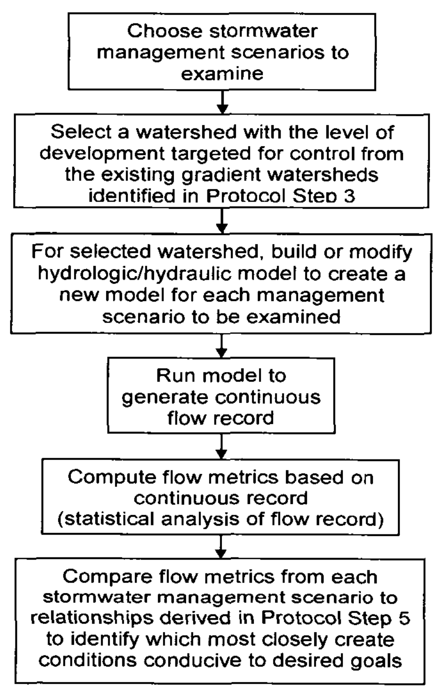


Figure 3-15. Basic Algorithm for Establishing Relationships Between Land Use Practices and Hydrogeomorphic Metrics.

hourly rainfall records from 1951 to 2000 were used to produce peak flow frequency exceedence and flow duration curves, as well as calculated values for the $T_{0.5}$ hydrologic metric. Stream channel erosion potential rates for a variety of critical shear stress values representative of both cohesive and non-cohesive sediments were also examined using the continuous flow record. Six different land use and runoff control scenarios were evaluated, and are described as follows:

- The Undeveloped watershed was simulated as a parcel of pastureland with five percent impervious area, where runoff was modeled to travel as overland flow to the outlet (Figure 3-16a).
- The Developed Uncontrolled watershed was modeled as a medium-density residential development with an average of thirty percent DCIA and a system of gutters, pipes, and swales directly connected to the outlet (Figure 3-16b) No stormwater controls were applied to the watershed in this scenario.
- In the Over Control scenario, runoff from the medium-density residential development described above was routed through controls designed to reduce the post-development 100-year peak flow rate to the pre-development 2-year flow rate. This practice is a stormwater management requirement for new development in Fort Collins and is intended to compensate for high-impact development that has already occurred by "over-controlling" runoff from all new developments.
- In the Over Control + Best Management Practices (BMP) scenario, over-control detention is combined with a water quality control BMP designed to

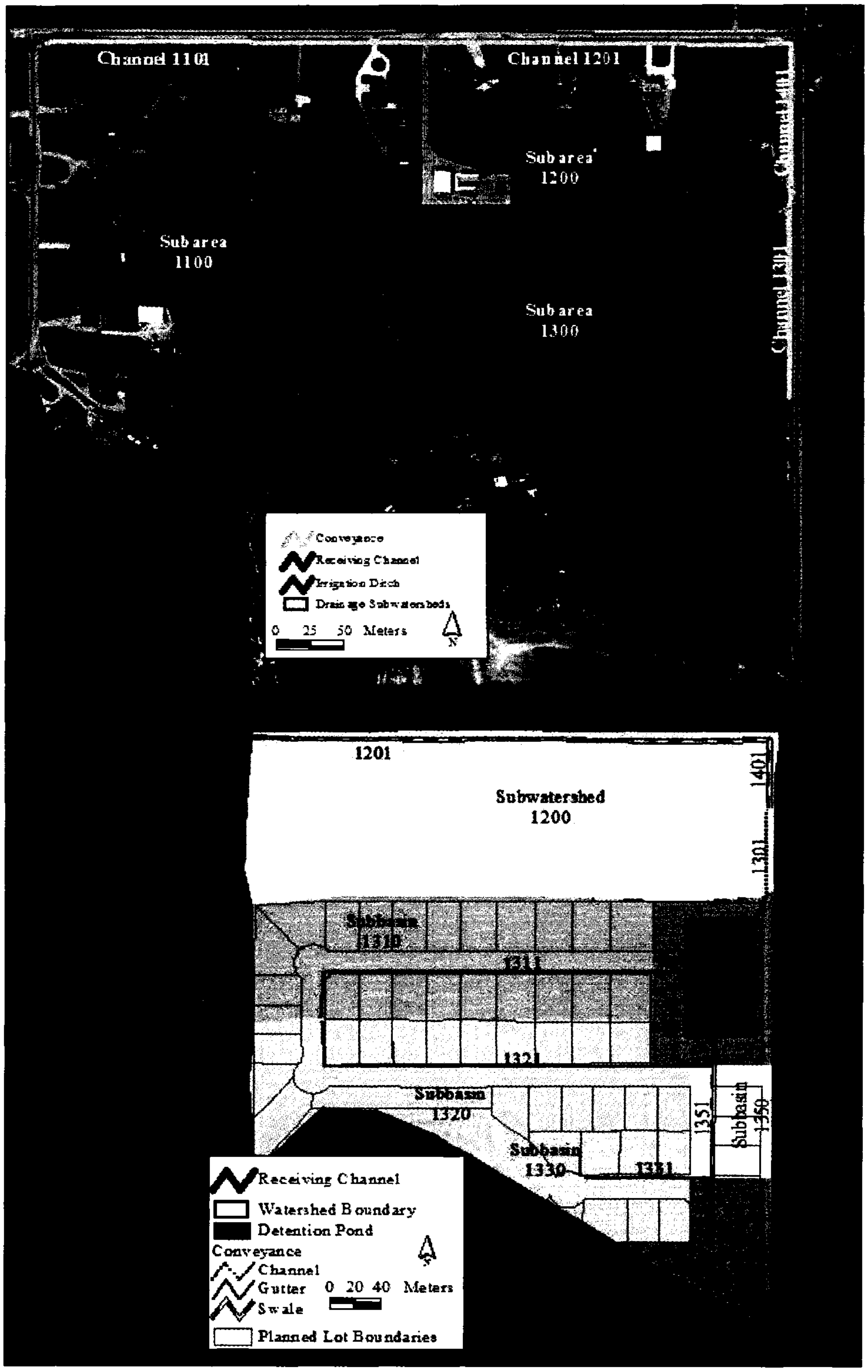


Figure 3-16. Hypothetical Development in Fort Collins, Colorado (a) Existing Conditions (b) Developed Conditions.

capture 70-90 percent of all runoff-producing events and have a draw-down time of 24 hours. These structures are designed to capture the "first flush" of runoff, which typically has the highest concentrations of pollutants. The water quality capture volume (WQCV) is the volume of water necessary to capture the "first flush" of runoff and is referred to as a BMP throughout the remainder of this section. The WQCV is intended to allow time for pollutants contained in the runoff to settle out before the detained water is discharged to the stream. The WQCV was calculated to be 0.041 ha-m, using the Denver Urban Drainage and Flood Control District (UDFCD) (2001) procedure.

- Two Peak Shaving + BMP control scenarios were evaluated using the standard practice of reducing the post-development peak flow rates to pre-development flow rates: (1) peak shaving of the 100-year storm; and, (2) peak shaving of the 100- and 10-year storms. Both of these scenarios were combined with a water quality BMP.

Response of Peak Flow Frequency Exceedence and Flow Duration Curves to Alternative Development Scenarios

Figure 3-17 shows the peak flow frequency exceedence curves for the five alternative stormwater management scenarios for the hypothetical development in Fort Collins, Colorado. In this figure the Peak Shaving + BMP scenario includes control of the 100- and 10-year storms. The effects of the over control scenarios are evident in the large reduction of peaks that occur infrequently. In the Peak Shaving + BMP scenario the peak flow frequency exceedence curve is well matched to undeveloped conditions for the 100-

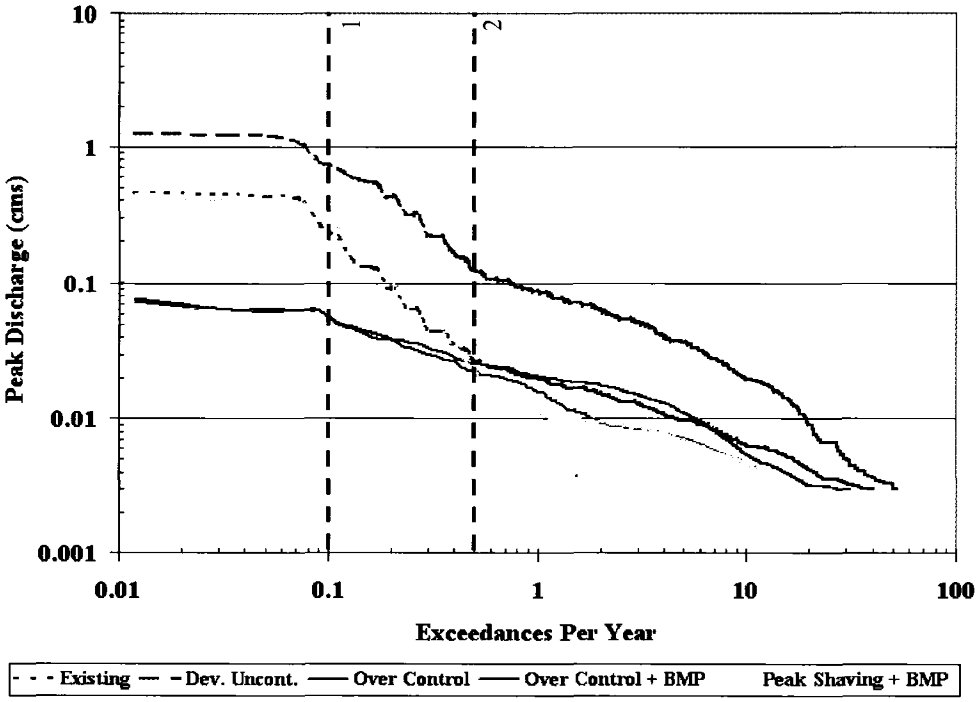


Figure 3-17. Peak Flow Frequency Exceedence in Fort Collins Watershed.

through the 2.5-year design storms, but peaks are over controlled for smaller storms due to the design of the WQCV outlet based on UDFCD (2001) design standards.

Flow duration curves are also useful tools in evaluating the impacts of stormwater management practices. However, simultaneous post-development reproduction of pre-development peak flow frequency exceedence and flow duration curves is not possible when stormwater detention and retention practices are used. These practices control the magnitude, timing, and duration of flows, but do not control the total volume of stormwater runoff. Volumes of stormwater runoff increase when development takes place, and these increased volumes are controlled by changes in the ability of the watershed to infiltrate or evapotranspirate as much water.

Figure 3-18 displays the percent of time discharges are equalled or exceeded over the 50-year simulation for undeveloped conditions and each post-development level of stormwater control examined. In this figure, 1 % represents one percent of the 50-year simulation period, or approximately 183 days; 10 % represents 1826 days (five years); 0.1 %, 18.3 days, etc. Figure 3-18 shows that when development takes place without any type of stormwater control, the discharges are greater in magnitude than "undeveloped" conditions for approximately 5.2 % of the overall time period examined, which is an average of 19 days/year. The duration of discharges in the "Peak Shaving (100 & 10) + BMP" scenario are less than those in the "Undeveloped" scenario between 0.43 cms and 0.26 cms, which is the maximum discharge rate for the 10-year storm. "Overcontrol" discharges are less than "Undeveloped" discharges for approximately 0.75 days/year.

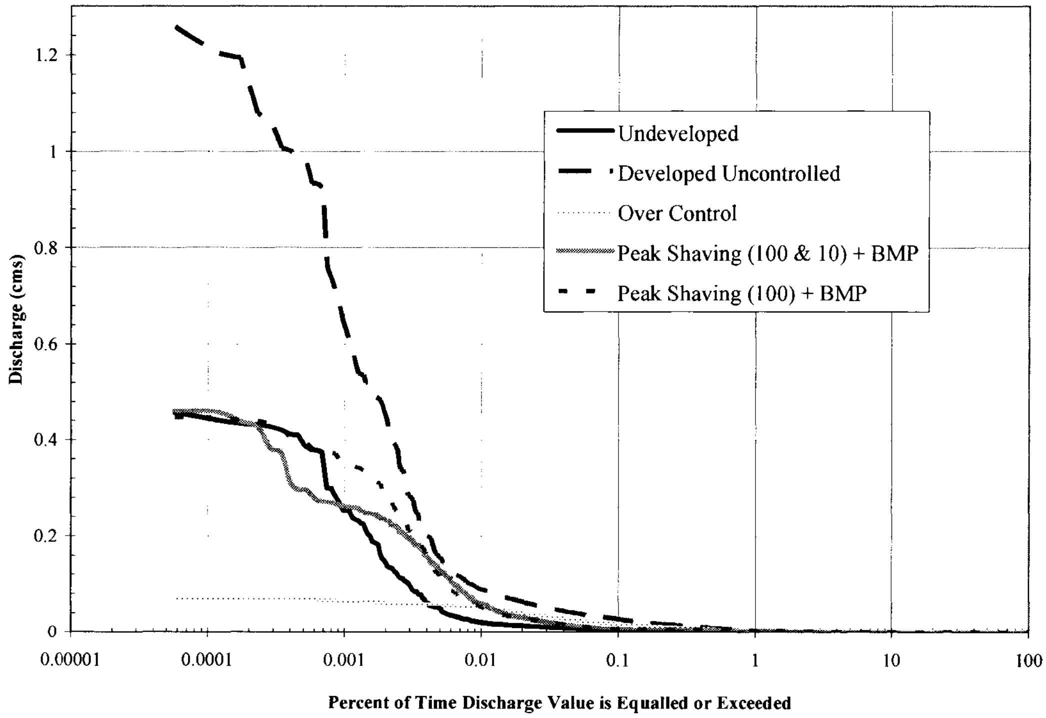


Figure 3-18. Flow Duration Curve for Fort Collins Watershed.

Effect of Alternate Development Scenarios on Erosion Potential for a Variety of Stream Bed Material Sizes

Inability to match the full duration curve does not mean that erosion potential due to changes in stream flow has to increase, so long as the critical portion of the flow duration curve is matched. The critical portion of the flow duration curve can be described as the duration of flow magnitudes above which stream bed or bank erosion occurs in the channel. If the duration of flows above the critical threshold remains the same from pre- to post-development, no increase in erosion potential takes place. It is important to note, however, that changes in sediment supply to the stream channel, such as those that occur due to development, will also affect the potential for erosion in the stream channel.

The critical portion of the flow duration curve varies depending on the critical shear stress of the stream bed and bank materials to be protected. In channels with low sediment entrainment thresholds, e.g., sands, fine gravels, and highly erodible clays with critical shear stresses less than one Pa, the rate of discharge required to entrain particles is so small that the flow duration curve should be controlled in its entirety. For stream bed and bank materials with higher entrainment thresholds, e.g., gravels and resistant clays, the critical discharge value, Q_c, can be estimated by first identifying τ_c; then using the shear stress equation to calculate the hydraulic radius value necessary to initiate particle movement; and lastly, using the Manning equation to calculate the critical discharge value. The duration of post-development discharges above the critical value should be restricted to the duration of the pre-development discharge. For example, if τ_c for the stream bed materials in the Fort Collins watershed is seven Pa, the y_c is 0.29 m, Q_c is

0.167 cms and the flow duration curve above this value should be matched to the extent possible.

Using the methods described in the Section 3.6.2.4, stream channel erosion potential was evaluated for a variety of critical shear stress values, ranging from 0.2 Pa to 10 Pa, and are presented in Figure 3-19 in which, erosion potential values less than one indicate a decrease in erosion rates pre- to post-development, and erosion potential values greater than one indicate an increase in erosion rates. Erosion potential values equal to one indicate no change in erosion rates. Figure 3-19 demonstrates what was difficult to discern in Figure 3-18: that the use of stormwater detention increases durations of the discharges that entrain sediment with shear stress values less than 0.7 Pa. When τ_c is between 0.2 and 0.7 Pa, the calculated erosion potential is larger when stormwater

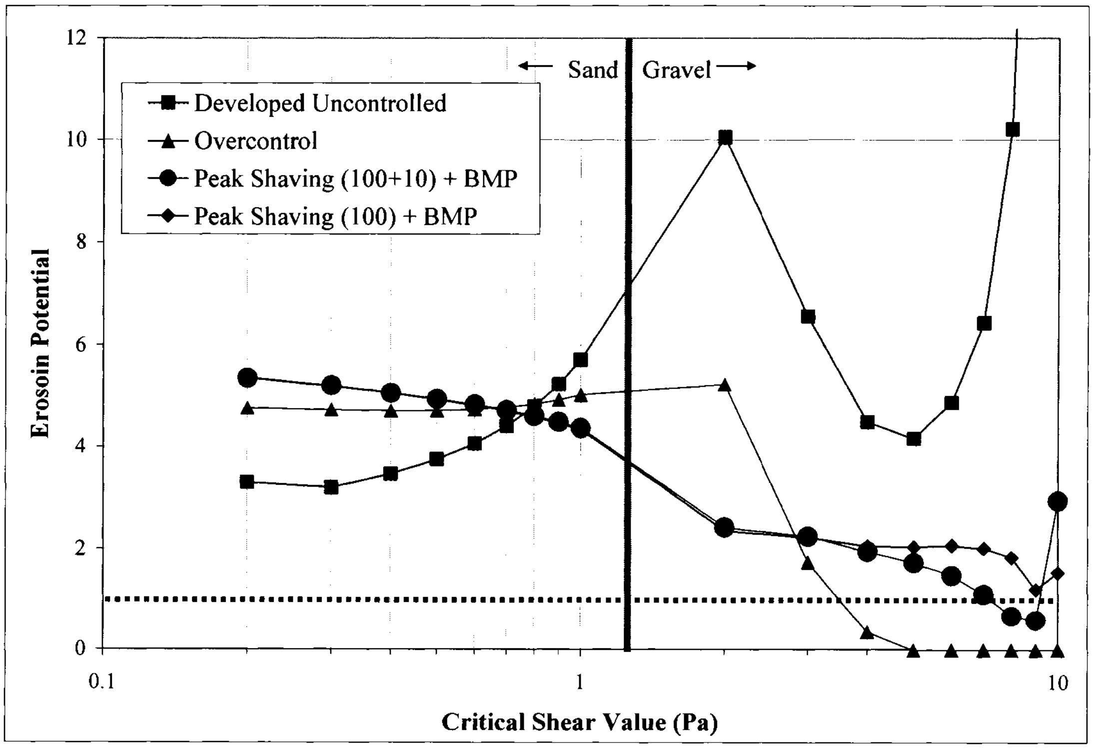


Figure 3-19. Erosion Potential at Varied Levels of Critical Shear Stress.

controls are in place than when stormwater is left uncontrolled. These results indicate that where stream channels consist of sand beds, detention could exacerbate erosion in a stream channel above what might occur if the stormwater were left uncontrolled. This increase in erosion potential suggests that a reduction in the total volume of runoff, through the use of infiltration, for example, may provide a better method for stormwater management for watersheds that drain to fine grained non-cohesive or erodible clay streams.

Figure 3-19 also shows that as the τ_c values increase, stormwater management practices create erosion potentials closer to the ideal value of 1.0. The erosion potential values of zero for the "Overcontrol" scenario demonstrate that no erosion is predicted under this scenario if the τ_c is greater than 4. Using the example of τ_c equal to seven Pa, Figure 3-19 shows that the "Peak Shaving (100+10) + BMP" scenario results in an erosion potential value closest to one, indicating that this type of stormwater control may be most appropriate to minimize changes in stream erosion potential when urbanization of the watershed takes place.

Erosion potential in the study reach was also evaluated for a grain size distribution consisting of d_{16}=0.505 mm (coarse sand), d_{50}=3.29 mm (very fine gravel), and d_{84}=12.3 mm (medium gravel). Erosion potential was calculated for individual grain sizes, and for the composite grain size distribution. For the composite grain size distribution analysis, average erosion rates for each time step were calculated using the d_{16}, d_{50}, and d_{84} values as well as estimates of the d_{33} and d_{57} values obtained through interpolation. Table 3-4 summarizes the erosion potential index calculated for each grain size, as well as for the composite grain size distribution. The erosion potential values for

Table 3-4. Erosion Potential for Representative Grain Size Distribution.

Scenario	d_{16} = 0.51 mm τ_c = 0.27 Pa	d_{50} = 3.3 mm τ_c = 2.2 Pa	d_{84} = 12.3 mm τ_c = 9.1 Pa	Full Distribution
Developed Uncontrolled	3.12	9.71	24.54	3.86
Overcontrol	4.75	4.5	0.00	4.80
Peak Shaving (100+10) + BMP	5.24	2.35	0.60	4.98
Peak Shaving (100) + BMP	5.24	2.28	1.14	4.96

each diameter correspond to those for similar τ_c values presented in Figure 3-19. This table shows that as τ_c values increase, the erosion potential increases when stormwater is left uncontrolled. Conversely, as τ_c values increase, erosion potential decreases when stormwater controls are implemented. When the grain size distribution is considered as a whole, the erosion potential is smallest when stormwater is left uncontrolled, similar to the differences between stormwater management scenarios observed for the d_{16} grain size. This is because of the significant contribution the smaller grain sizes make to the overall erosion rate.

These results show that erosion index values are highly sensitive to the critical shear stress values required to initiate erosion. Selection of the stormwater controls best suited to minimize changes in erosion potential requires the identification of these critical shear stress values. Identification of these critical shear stress values requires field evaluation using sieve analysis or direct measurement of erodibility using methods such as jet-testing.

Response of $T_{0.5}$ Hydrologic Metric to Alternative Development Scenarios

Figure 3-20 shows the response of the modified $T_{0.5}$ hydrologic metric to urban development with varying levels of controls (Edgerly, 2006; Edgerly et al. 2006). In this

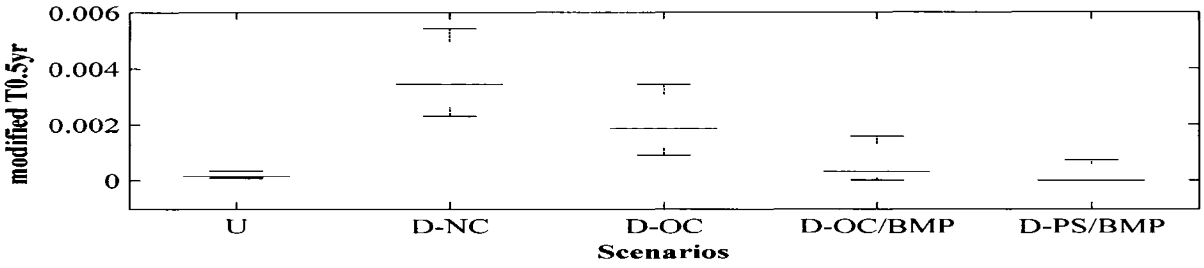


Figure 3-20. Median and Interquartile Range Values of Modified $T_{0.5}$ for Multiple Scenarios of Development. U=Undeveloped, D-NC=developed, uncontrolled, D-OC=Overcontrol, D-OC/BMP: Overcontrol + BMP, D-PS/BMP: Peak Shaving (100 & 10) + BMP (Source: Edgerly, 2006).

analysis, the modified version of the $T_{0.5}$ metric was computed using a fixed pre-development 0.5-year peak flow rate to standardize flow duration curves. This approach is consistent with that used for metrics computed by Cassin et al. (2005) in the King County Normative Flows Project: the T_{Qmean} and the percent of time above 2-year mean flow. As expected, the controls were most effective at mitigating the effects of urban development on modified $T_{0.5}$, since they were designed to prevent post-development flow rates from exceeding the target flow rates. Peak shaving + BMP controls also appear to bring this metric closer to pre-development levels than the other designs. This makes sense because control of the water quality capture volume (BMP) limits the range of possible flows by controlling smaller storms which are closer in size to the half-year storm. Also, by reducing the 100-year peak to the pre-development 2-year peak discharge, the over-control designs increase the durations of intermediate flows by releasing the excess volumes at lower rates for longer durations than the peak shaving designs.

3.8.2.2 Impact of Runoff Control Strategies on Geomorphic Metrics in Lenexa, Kansas

Geomorphic metrics were used in Lenexa, Kansas to develop design criteria for detention basins so that the current geomorphic condition in the natural stream system would be preserved to the maximum extent practicable (Rohrer et. al., 2005; Pomeroy et al., *in press* a). Fifty-three years of historical precipitation data were applied to the SWMM4.4h model to conduct continuous hydrologic and hydraulic simulations of a portion of the Cedar Creek watershed in Lenexa, Kansas, generating continuous flow hydrographs in the receiving stream channels.

The study area is 93 hectares and is currently undeveloped, consisting mostly of pasture land and natural prairie. Development plans for the watershed include the addition of a business park and medium density residential neighborhoods that will increase total impervious area (approximately two percent, currently) to an average of 57 %. The study area was divided into 12 subbasins, as shown in Figure 3-21.

A number of levels of stormwater detention were examined. Peak shaving was applied for various combinations of the one-, two-, and 10-year design storm events. The application of stormwater BMPs was also examined. Again, the term BMP represents control of the water quality capture volume (WQCV), which in this case was computed as the product of the 90% rainfall depth and a runoff factor based on percent impervious using the method described by the Mid-America Regional Council (MARC) and American Public Works Association Manual (APWA) for Best Management Practices for Stormwater Quality (2003). Varied BMP draw down times of 24-48 hours were also examined. Results for the following three scenarios are described in the remainder of this section:

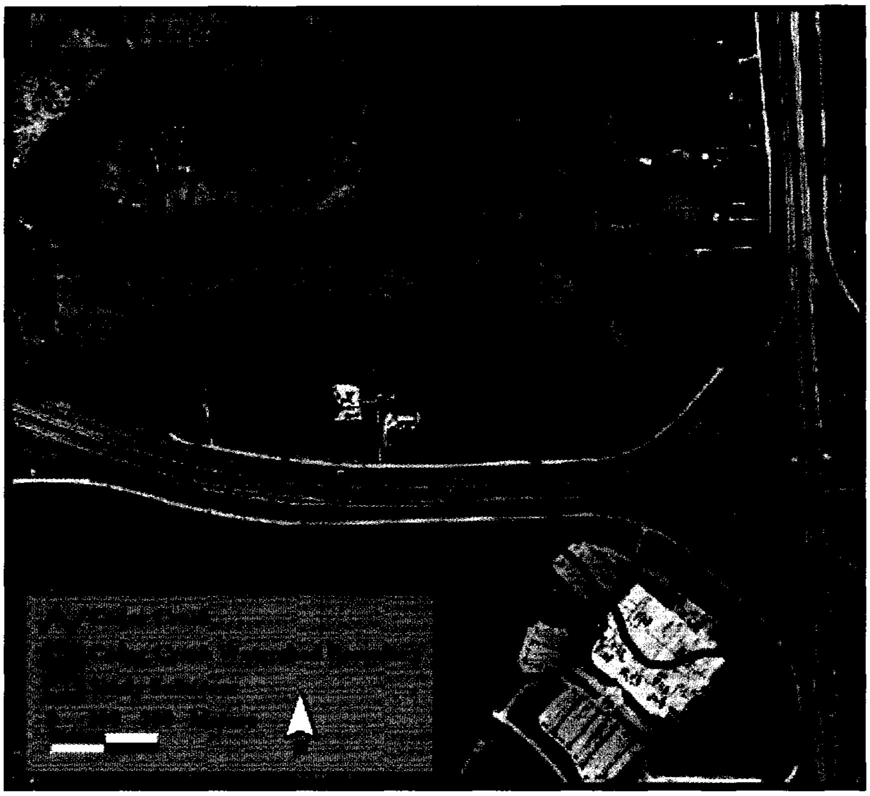


Figure 3-21. Cedar Creek Watershed Study Area.

1. "Undeveloped": In this scenario all subbasins in the study area are natural, with an estimated imperviousness of two percent. This scenario assumes the watershed is drained by natural channels.

2. "Developed Uncontrolled": This scenario is based on future land use conditions (residential and commercial) with a typical stormwater conveyance system consisting of concrete pipes that drain to the natural stream system.

3. "Developed, Controlled": This scenario includes peak shaving of the one- and 10-year storm and control of the WQCV using extended detention with a 40 hour draw-down time. This scenario represents on-site BMP and regional detention criteria that were selected to allow post-development replication of

pre-development flow frequency curves and the critical portions of the shear stress duration curves.

Instream continuous stage data generated by the SWMM4.4h model were used to examine stream channel erosion potential through the use of the erosion potential index described in Section 3.6.2.4. Critical flow thresholds were extrapolated from critical shear stress values by using the shear stress equation to calculate the hydraulic radius value necessary to initiate particle movement; and then using the Manning equation to calculate the critical discharge value. These critical flow thresholds were used to identify return frequencies above which the duration of stormwater must be controlled to pre-development levels.

In-stream critical shear stress values were identified during field surveys of the upper tributaries and the main stem of Cedar Creek. Table 3-5 summarizes the soil types observed and the literature reference used to identify critical shear stress values for each location based on observations of soil types and channel conditions in the field (Kwiatkowski, 2004).

Figure 3-22 displays the percent of time in-channel shear stresses are equaled or exceeded over the 53-year simulation for "undeveloped" conditions, "developed, uncontrolled" conditions, and the recommended "developed, controlled" conditions.

Table 3-5. Summary of Critical Shear Stress Values and Soil Types in the Cedar Creek Watershed.

τ_c (Pa)	Stream Location	Representative Soil	Reference
12.4	Upper Trib.	colloidal alluvial silts	Chow, 1959
38.3	Main Stem	graded silts to cobbles	USDA-ARS-NSL, 2001

Source: Kwiatkowski, 2004.

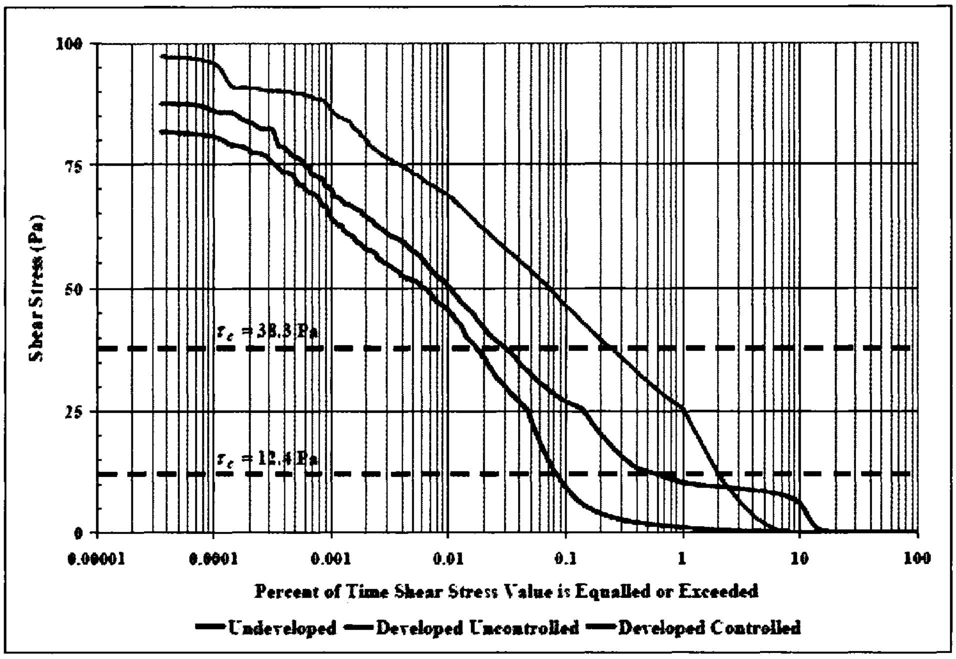


Figure 3-22. Shear Stress Duration in Cedar Creek Watershed

Figure 3-22 shows that when development takes place without any type of stormwater control, the "developed, uncontrolled" shear stress values are greater in magnitude than "undeveloped" conditions for approximately 10 % of the time, or an average of 37 days/year. The magnitude and duration of shear stresses in the "developed, controlled" scenario are greater in magnitude than "undeveloped" conditions for approximately 19 % of the time (average of 68 days/year).

The τ_c values determined for Cedar Creek are shown on Figure 3-22, and shear stresses above these critical values were those targeted for control. This figure shows that the duration of time the τ_c value for the main stem (38.3 Pa) is equaled or exceeded increases by an order of magnitude between "undeveloped" and "developed, uncontrolled" conditions. In "undeveloped" conditions, the τ_c value for the main stem (38.3 Pa) is equaled or exceeded 0.02 % of the time (three days), and in "developed, uncontrolled" conditions, τ_c is equaled or exceeded 0.23 % of the time (45 days). The "developed, controlled" condition allows a closer duration match to "undeveloped" conditions, with the critical shear stress threshold τ_c equaled or exceeded 0.03 % of the time, or six days. Figure 3-22 also shows that if the τ_c value for the reach were a lower value, such as the value determined for the upper tributaries, 12.4 Pa; the critical portion of the shear stress duration curve would not be as well matched. For this critical threshold, the duration of time τ_c is equaled or exceeded ranges from 0.08 % (average of one third of one day/year) in "undeveloped" conditions, 0.54 % (average of 2 days/year) in "developed, controlled" conditions, and 2.10 % (average of 8 days/year) in "developed, uncontrolled" conditions. The order of magnitude increases in the duration of time at this lower critical shear stress value, regardless of the stormwater control, demonstrates the importance of identifying the critical shear stress threshold when designing stormwater management practices to minimize stream channel erosion potential.

Table 3-6 quantitatively describes differences in excess shear using the stream channel erosion potential index discussed in Section 3.6.2.4. Table 3-6 shows that for both τ_c values, erosion potential is decreased with the application of the "developed, controlled" scenario that includes peak shaving of the one- and ten-year storm and control of the WQCV using extended detention with a 40-hour drawdown.

Table 3-6. Stream Channel Erosion Potential Under Developed Conditions.

Scenario	Erosion Potential	
	τ_c=12.4 Pa	τ_c =38.3 Pa
Developed, Uncontrolled	19.0	13.0
Developed, Controlled	2.7	1.8

Critical flow thresholds were used to identify return frequencies above which the flow duration curve must be controlled to pre-development levels. The return periods for these critical flow thresholds are shown in Figure 3-23. It should be noted that because shear stress is dependent upon channel geometry, the critical discharges presented in Figure 3-23 are for the downstream-most channel in the study area, only. Figure 3-23 shows that the flow frequency curve for the "developed, controlled" conditions approximately matches the flow frequency curve for the "undeveloped" conditions, as designed, for the one- and 10-year return interval design storms. This match of the flow frequency curve, however, does not guarantee a match in the duration of critical flows, as can be seen from the previous evaluation of the shear stress duration curves.

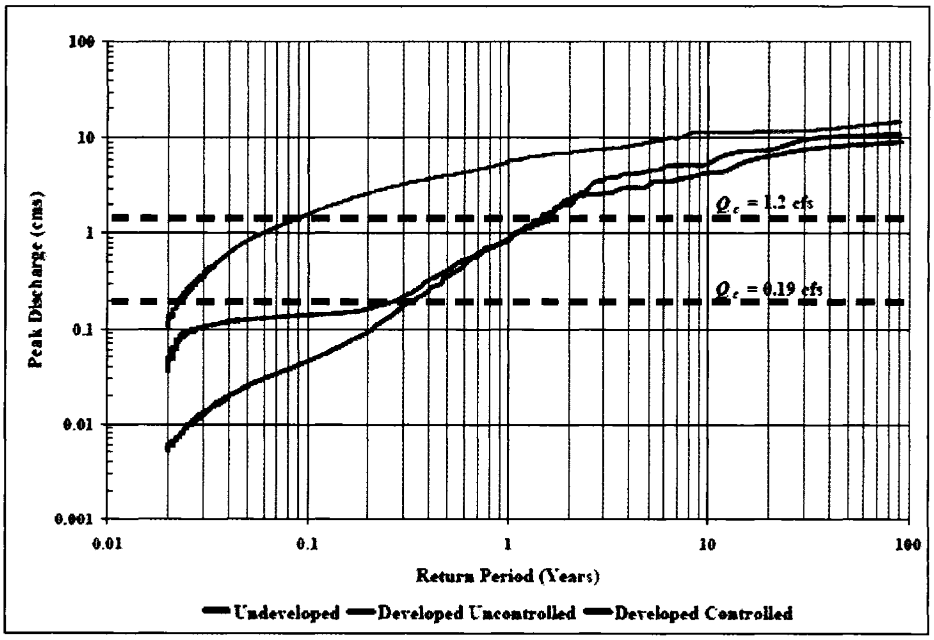


Figure 3-23. Peak Flow Frequency Exceedence Curve for Cedar Creek watershed.

3.9 Step 7 - Establish Management Criteria for Stream Types of Interest

This step (Figure 3-24) is the institutional or administrative process of setting the stormwater management criteria, basically the design criteria for runoff controls on new development. The rules, which will be based on the analyses recommended in this document, should be straight forward and understandable by the average consulting engineering firms that do development work. Criteria can be set by municipal administrators based on analysis of relationships established by the protocol described in the previous sections. Stormwater management practices can be evaluated to identify those which provide hydrologic or geomorphic metric results in the targeted range. An example of this comparison is illustrated in Figure 3-25.

Figure 3-25 shows the regression of qualitative multihabitat EPT richness data collected by the USGS in eight watersheds in the North Carolina piedmont and the T_1 hydrologic metrics computed from 20-year SWMM simulations for the same watersheds. The T_1 hydrologic metric is computed as the percent of time the flow is above the 1-year return interval peak discharge computed for each watershed for the period of record examined (20-years, in this example). Qualitative multihabitat samples are those collected from multiple types of habitat in a stream, as opposed to samples collected only in riffles, which provide a rich habitat environment for macroinvertebrates. Further details on the biologic data collected by the USGS in the North Carolina piedmont and computation of the T_1 hydrologic metric are given in Section 4.0.

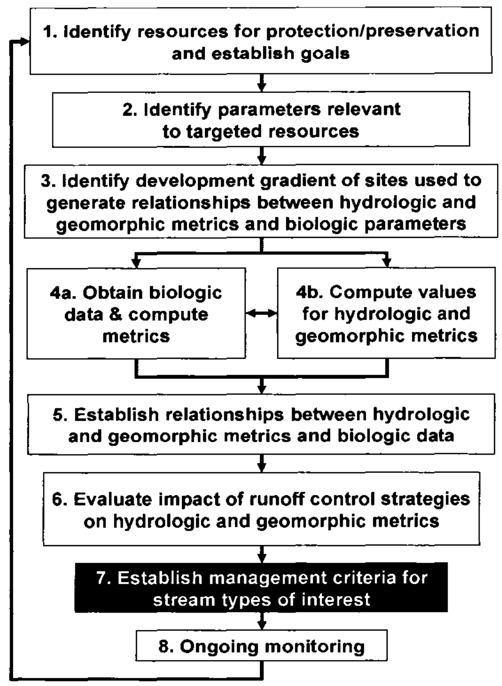


Figure 3-24. Establish Management Criteria.

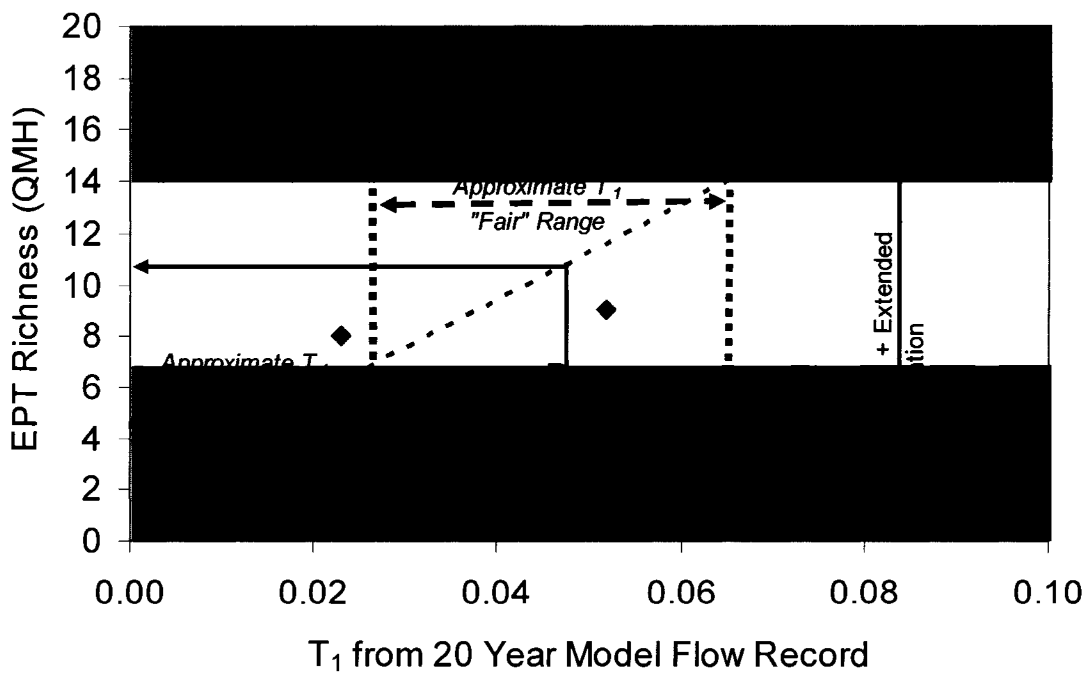


Figure 3-25. Evaluating the Effects of Stormwater Management Criteria on Expected EPT Richness.

The regression relationship shown in Figure 3-25 can be used to identify approximate ranges where the T_1 metric allows for various levels of EPT richness scores. It is important to note that the ranges are not thresholds and that there is a band of uncertainty associated with the dashed lines that separate the "Poor", "Fair" and "Good-Fair" ranges.

The T_1 or other hydrologic metrics can be computed from rainfall-runoff model scenarios used to evaluate various stormwater management practices, as demonstrated in Section 3.8.2.1. The hydrologic metric values can be compared to regression results to identify the biologic score expected if the stormwater management practice were implemented. Hypothetical examples of the effects of peak shaving detention, peak shaving and extended detention, and the effects of no stormwater controls are shown in Figure 3-25. In this example, a combination of peak shaving and extended detention allow for a "good-fair" EPT richness score. Examples of the effects of peak shaving and extended detention on the $T_{0.5}$ hydrologic metric in the Morgan Creek watershed in the North Carolina piedmont are given in Section 5.3.

Results from application of the protocol in the North Carolina piedmont presented in sections 5.2.1 and 5.2.2 showed that macroinvertebrate metrics are related to both hydrologic metrics and substrate size as measured by the predominant size class found in each watershed. Further relationships show that the predominant size class found in each watershed is correlated to the $T_{0.5}$ hydrologic metric, indicating that these two variables are not independent. It is important to note that achievement of a hydrologic metric in the desired range may not produce satisfactory biotic results if other factors (such as substrate size) are negatively influencing biotic habitat. Minimization of changes to hydrologic metrics, however, will likely reduce changes in bed substrate critical for macroinvertebrate habitat and is likely to be beneficial in developing watersheds because it may reduce changes in bed substrate, but is likely to be less beneficial in restoration efforts of watersheds that are developed and have already experienced changes in bed substrate due to hydrologic alteration. Hydrologic metric can be used as mechanistic links between urbanization and biotic response, but may not be stand alone linkages since bed substrate is also critical linkage between urbanization and biotic response.

In all the applications investigated by the CSU study team, which include moderate rainfall areas like Eastern Kansas and semi-arid areas like Fort Collins, Colorado, it appears that if drainage criteria equivalent to the following are required for developing watersheds, that the negative hydrologic and geomorphic impacts due to urbanization will be reduced, albeit not zero:

- Control of the peak runoff from the 100- and 10-year post-development flows to the peak flows in the pre-development condition
- Control of the post-development peak runoff from a small storm with return interval in the range of 1 - 2 years to the peak flow in the predevelopment condition. The correct storm should be determined by the stormwater management agency by fitting the pre-development peak flow-frequency exceedence curve.
- BMPs that have equivalent hydraulic performance to that of an extended detention basin, sized in accordance with the Manual of Practice for Urban Stormwater Management (WEF and ASCE, 1998) and having a 40-hour drawdown time.

Many smaller jurisdictions may not have the resources to carry out the protocol as developed in this project. For such entities, it is recommended that these runoff control criteria provided above be used as a minimum standard.

It should be noted that volume controls, such as infiltration, were not examined in this study. Use of such devices can be evaluated using the protocol and may change some of the above recommendations for runoff control measures in developing watersheds. Also, low impact development (LID) techniques were not evaluated in this study. These controls provide volume reduction through infiltration, as well as flow attenuation, and should be evaluated to determine their relative impact on hydrologic and geomorphic metrics.

3.10 Step 8 - Ongoing Monitoring

Once the management criteria have been established and implemented in Step 7, Step 8 of the protocol calls for ongoing surveillance monitoring to evaluate the effectiveness of protection efforts. Given a biological condition gradient in which to frame further study observations, surveillance monitoring can focus on the evaluation of watersheds targeted for protection or restoration. Surveillance monitoring efforts are intended to evaluate biologic condition and the status of hydrologic and geomorphic parameters in the protected watersheds to determine whether management measures are adequately protective. The sampling protocols identified and discussed in the previous sections should generally be appropriate for ongoing surveillance monitoring, with an increased emphasis on the protected watersheds.

In the event that one or more hydrologic, geomorphic, or biologic parameters fails to meet established protection goals for the target watersheds, an adaptive management approach is recommended. Sampling parameters and associated protocols should be refined at that point for the purpose of identifying the stressor(s) resulting in non-attainment of biotic goals. In some cases, management criteria derived from the initial protocol implementation could prove ineffective against one or more environmental stressors. In this event, the protocol should be repeated for the refinement of biotic goals (Figure 3-26, Step 1), or refinement of selected measurement parameters and associated linkages (Figure 3-26, Steps 2 through 6). Management criteria might then be revised or replaced with alternative measures to more effectively address the identified stressors.

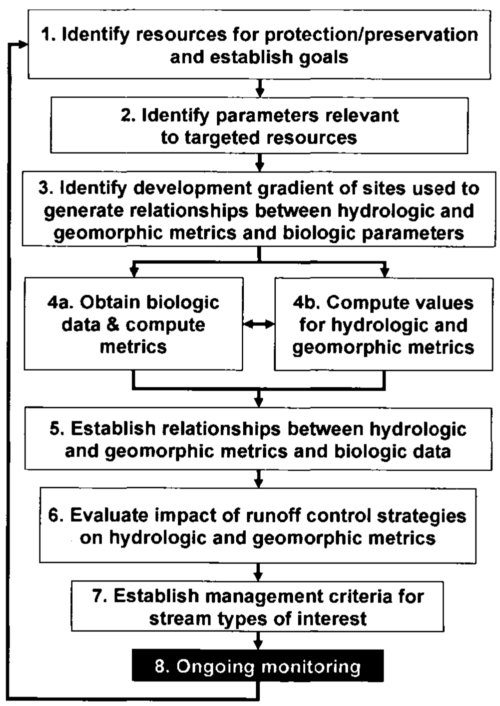


Figure 3-26. Ongoing Monitoring.

4.0 DATA AND MODEL DEVELOPMENT FOR WATERSHEDS IN THE NORTH CAROLINA PIEDMONT

4.1 Selection of Test Areas

The protocol was applied in the Raleigh, North Carolina area, using data gathered by the USGS under the National Water Quality Assessment (NAWQA) program. Specifically, data collected in the North Carolina Piedmont under the study Effects of Urbanization on Stream Ecosystems (EUSE) were used for application and testing of the protocol described herein. The objectives of the USGS EUSE study are to determine the physical, chemical, and biological responses of stream systems to a gradient of increasing urban intensity; determine the most important landscape features driving hydrologic, chemical, geomorphic, habitat, and biological responses to urbanization; determine the physical and chemical factors associated with biological responses; and compare these responses among metropolitan areas located in different geographic settings (climate, geology, hydrology, biology, water use) in 11 regions across the United States (USGS, 2007).

The Raleigh, North Carolina region was chosen for application and testing of the protocol based on the size of the study watersheds and the willingness of the USGS to share the data at the time it was needed for testing of the protocol. Nearly half of the 30 watersheds investigated by the USGS in the Raleigh, North Carolina region are less than

5 square miles, a size that is practical for modeling at the level of detail needed to produce data useful for calculating hydrologic and geomorphic metrics. This watershed size is also the typical size that a stormwater management agency would consider for application of the protocol. Additional factors which discouraged selection of alternate USGS study regions for application of the protocol included the influence of combined sewer overflows and the influence of transbasin diversions in arid and semiarid regions.

The 30 North Carolina watersheds were screened against the site selection criteria summarized in Section 3.5, including identification of watersheds with homogeneous land use (relatively uniform, rather than mixed land use within the watershed), streams with similar geomorphic characteristics, watershed size, and the elimination of watersheds with confounding factors such as point source discharges and dams. Watersheds were also selected based on the degree of urbanization in each basin as represented by multi-metric Urban Intensity Index (UII) values computed by the USGS (McMahon and Cuffney, 2000; Coles et al., 2004; Tate et al., 2005). The UII includes census, socioeconomic, infrastructure, land-use and land-cover metrics that correlated with population density. The variables evaluated include population density, household density, road density, percent of basin in urban land, and percent of buffer (90 m on either side of the stream) in urban land. Rankings of the UII variables were used to create an index specific to the 30 North Carolina watersheds that varied from zero to 100. Zero represent the least amount of urbanization in the 30 watersheds and 100 represents maximum urbanization within the set of 30 watersheds (Giddings et al., 2007).

Ten watersheds were initially selected for application of the protocol; these watersheds ranged in size from 3 - 9 square miles. The selection of several watersheds larger than the recommended size criterion of 5 square miles illustrates the nature of the above criteria as a recommended point of departure, and not absolute rules for site selection. Ultimately, two of the ten selected watersheds were eliminated from the protocol application. One was eliminated based on the recommendation of USGS NAWQA staff in North Carolina because of heavy development that took place within the watershed during the time of the EUSE study. The other watershed was eliminated was due to time constraints of CSU project team staff during field surveys. Table 4-1 lists the selected NAWQA study watersheds, their corresponding drainage area size, and their associated level of urbanization/development as represented by the UII. The location of these watersheds is shown in Figure 4-1.

Table 4-1. Characteristics of Watersheds Chosen for Protocol Application.

Watershed	USGS Site Number	Basin Size (sq mi.)	Urban Index (UII)	Total Impervious Area (%)	Homo-geneity (0=Low 1=Med 2=High)	Stream bed Material % Silt / Clay / Sand	% Gravel / Cobble
Morgan Creek	02097464	8.3	4.9	0.8	2	16.3	58.2
Strouds Creek	0208501535	8.9	7.2	3.8	1	3.6	81.9
Camp Branch	0208794025	3.2	12.9	2.9	1	16.4	78.1
Wilson Creek	0209750881	3.5	24.1	9.0	1	34.6	63.6
Dutchmans Branch	0208758440	4.6	28.6	4.4	1	9.1	90.9
Black Creek	0208725055	3.6	79.6	20.5	2	23.6	16.3
Beaverdam Creek	02087304	3.0	90.8	11.7	2	40.0	58.2
Pigeon House Creek	0208732610	4.5	100.0	30.6	2	30.9	58.2

4.2 Data Development and Watershed Model Calibration

4.2.1 Data Collection

Data measured by the USGS at the EUSE sites included physical stream characteristics such as physical habitat, stream stage, and water temperature; chemical

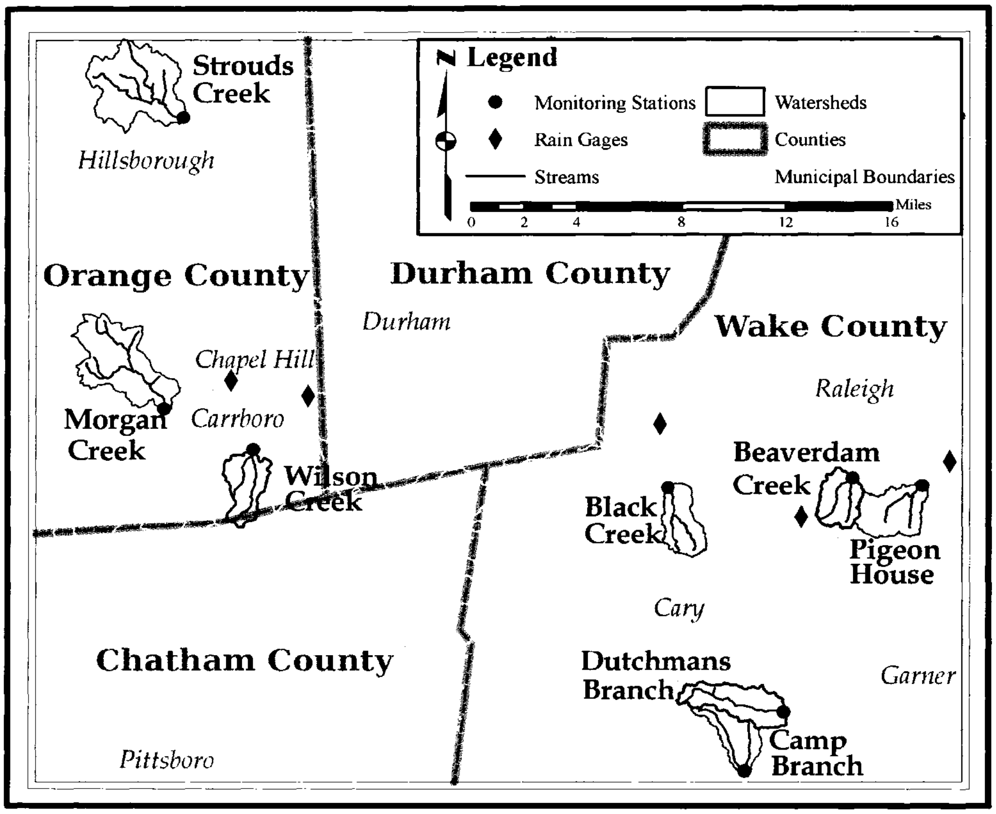


Figure 4-1. Study Watersheds.

constituents, including nutrients, pesticides, organic carbon, suspended sediment, sulfate, chloride, dissolved oxygen, pH, specific conductance in water, organic contaminants and potential toxicity; and biological integrity including invertebrate, algae, and fish communities, and algal biomass (USGS, 2007).

Benthic macroinvertebrate data were collected by the USGS in the North Carolina area from early May to late June 2003 using standard NAWQA program sampling protocols (Moulton et al., 2002). Macroinvertebrates were collected using two types of samples - a quantitative sample collected from instream riffles (richest targeted habitat, RTH) and a qualitative multihabitat (QMH) sample that collected invertebrates from as many habitats in the stream reach as were accessible (Giddings et al., 2007). Figure 4-2 shows richness values for Ephemeroptera, Plecoptera, and Trichoptera (EPT) for the RTH for the 30 watersheds in the North Carolina Piedmont, and shows that EPT richness varies with UII. Ratings for various EPT richness scores are given in Table 4-2. It should be noted that only one of 30 samples generated EPT richness rating of good-fair, and the remainder were fair or poor. This lack of a full range of biotic integrity is perhaps due to historic land use within the North Carolina Piedmont; while a number of the undeveloped watersheds in the study area were forested or partially-forested at the time of the study; it is likely that these watersheds were historically used for agriculture.

Bed substrate composition was evaluated by the USGS in the EUSE watersheds using pebble count methods described by Fitzpatrick et al. (1998). Characteristics at the

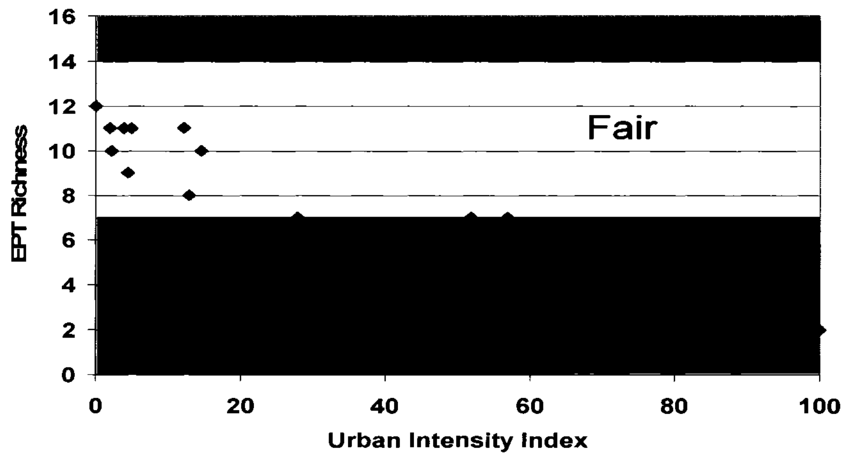


Figure 4-2. Range of EPT Richness Scores in North Carolina Area.

Table 4-2. EPT Richness Rating Scores.

Rating	EPT Richness
Excellent	>27
Good	21-27
Good-Fair	14-20
Fair	7-13
Poor	0-6

Source: NCDENR, 2006.

reach scale were measured at 11 equally spaced transects along the sampling reaches (USGS, 2007), and five samples were evaluated at each transect. Table 4-3 summarizes the bed substrate data collected by the USGS, and shows the percent of samples with the dominant substrate in each size class for each watershed.

Table 4-3. Summary of Dominant Bed Substrate in the Test Watersheds.

Class Name	d_s (mm)	Strouds	Morgan	Camp	Wilson	Dutch-mans	Black	Beaver-dam	Pigeon
Silt/clay/marl/muck/ organic detritus	-	4%	13%	9%	7%	6%	15%	6%	0%
Sand	>0.063	0%	4%	7%	27%	4%	9%	35%	31%
Fine/medium gravel	>2	18%	4%	13%	15%	36%	2%	22%	22%
Coarse gravel	>16	6%	6%	35%	26%	22%	4%	18%	20%
Very coarse gravel	>32	11%	22%	20%	20%	16%	2%	16%	6%
Small cobble	>64	22%	22%	9%	4%	15%	4%	0%	7%
Large cobble	>128	26%	6%	2%	0%	2%	6%	2%	4%
Small boulder	>256	11%	7%	2%	0%	0%	7%	0%	2%
Other	-	4%[1]	18%[1]	4%[2]	2%[1]	0%	53%[1]	2%[1]	9%[1]

[1]Large boulder, irregular bedrock, irregular hardpan, or irregular artificial surface, $d_s > 512$ mm
[2]Smooth bedrock, concrete, or hardpan.

Stream depth data were collected by the USGS at the outlet of each of the EUSE watersheds using pressure transducers. Water level data were collected at five-minute increments for a portion of the monitoring period, and at 15-minute increments for the remainder of the monitoring period. For the eight watersheds applied in the protocol, the period of record varied from 12 months to 17 months, depending on the monitoring station. Three of the eight stations contained consecutive data gaps of more than one day, and these are indicated in Table 4-4. The period of record for each station is summarized in Table 4-4. The water level data were converted to discharge values by USGS staff using stage-discharge rating curves established using HEC-RAS models (Giddings, 2006). Computed discharge values were not provided by the USGS for Strouds Creek due

Table 4-4. Dates of Water Level Monitoring Data.

Watershed	USGS Site Number	Length of Record (months)	Range of 5-Minute Data		Range of 15-Minute Data		Data Gaps (days)
Morgan Creek	2097464	17.6	N.A	N.A.	15JUN02	30NOV03	-
Strouds Creek	208501535	15.2	12JUL02	16DEC02	16DEC02	19OCT03	20
Camp Branch	208794025	15.7	30JUL02	18DEC02	18DEC02	19NOV03	-
Wilson Creek	209750881	15.6	31JUL02	16DEC02	16DEC02	19NOV03	-
Dutchmans Branch	208758440	15.7	30JUL02	18DEC02	18DEC02	19NOV03	-
Black Creek	208725055	16.7	01JUL02	16DEC02	16DEC02	21NOV03	-
Beaverdam Creek	2087304	12.6	27JUN02	17DEC02	17DEC02	05AUG03	20
Pigeon House Creek	208732610	15.1	01JUL02	17DEC02	17DEC02	19NOV03	47

to problems with measured cross-section data and the HEC-RAS model for that watershed (Giddings, 2005).

Field data were also collected by the CSU project team. In November of 2005, stream cross-section data were measured using level and rod methods described by Harrelson et al. (1994). Locations of the cross-sections in the eight watersheds were recorded by Garmin *e*trex global positioning system (GPS) units. Stream cross-sections were measured at approximately 2500-foot intervals along stream channels within each of the watersheds. In addition to collecting cross-section data, in-channel Manning roughness values were identified using guidelines provided by Chow (1959) and Arcement and Schneider (1989).

Precipitation data were obtained for application to the SWMM5 models. Locations of the rain gages considered for use are shown in Figure 4-1. Five rain gages were identified in the proximity of the study watersheds with long-term (greater than five years) data. Of these gages, two had 15-minute data and the others had hourly rainfall data. Two of the gages did not have a desired record length of 20 years, and one other had large gaps in its record. Annual precipitation summaries for the two gages with adequate rainfall records are shown in Table 4-5. One gage, monitored by the National Weather Service (NWS) at the Raleigh Durham International Airport (Coop ID 317069), has hourly rainfall records; the other monitored by the USGS at Marsh Creek near New

Table 4-5. Annual Precipitation Summaries.

Year	National Weather Service Raleigh Durham International Airport Coop ID 317069 (inches)	USGS Marsh Creek near New Hope, NC Site No: 0208732885 (inches)
1986	36.95	42.37
1987	42.09	56.03*
1988	37.66	52.44
1989	54.21	55.74*
1990	37.55	39.2*
1991	35.46	36.16*
1992	43.18	43.17*
1993	38.05	39.29
1994	36.41	35.89*
1995	48.59	56.84*
1996	45.02	41.7*
1997	40.84	27.81*
1998	50.14	44.89
1999	50.00	57.26
2000	37.07	39.53*
2001	33.77	28.56
2002	45.98	42.43
2003	47.45	45.13
2004	46.00	33.50
2005	37.53	35.26
Mean	42.20	43.02

*Indicates that the record for that year was revised or supplemented by CSU personnel with data from the National Weather Service gage at Raleigh Durham International Airport Coop ID 317069.

Hope, NC (Site No 0208732885), has 15-minute rainfall totals. As can be seen, the annual totals for each year are similar, and the overall average of the totals is nearly the same.

The USGS rainfall record at Marsh Creek near New Hope, NC (Site No 0208732885) contained 13 data gaps of 30 days or longer. Data from NWS gage at the Raleigh Durham International Airport (Coop ID 317069) were examined for the periods of record during these data gaps to determine if it was likely that these data gaps were dry periods, or if the USGS gage was not functioning properly during the data gap time periods. Based on this analysis, 11 time periods ranging from 31 to 643 days were identified and filled in with data from the NWS gage. The hourly data from the NWS gage were translated to 15-minute data using NetStorm (Heineman, 2007) and combined with the USGS gage data to form a complete rainfall record. Additionally, the precipitation data from the USGS gage were evaluated to identify those values that seemed out of the normal range of expected values. All records of one inch or more were individually examined, and several were modified based on values from the NWS gage, or were deleted altogether.

4.2.2 Model Development

Guidelines provided in Step 4b of the protocol were used to develop SWMM5 models using both geographic information system (GIS) and field-determined data (see Section 3.6.2.1). Each watershed was discritized into subcatchments by applying the ArcHydro extension of ArcGIS to 10-m digital elevation model (DEM) data from the USGS Seamless website (USGS, 2006a.) Hydrologic characteristics for subareas in each watershed were estimated by area weighting measurements of slope, runoff length, soil type and imperviousness. This section defines the model parameters that were used in SWMM5 and identifies the basic calibration technique used to determine their values; details of the calibration algorithm used in this study are provided in Section 4.2.3.

The first step in model development was to lay out the stream system to be modeled for each watershed, then divide the watershed into smaller catchments (subareas) that drain to the stream system. Criteria for defining a catchment were based on guidance for SWMM application by Huber and Dickinson (1998) and included uniform catchment slope, surface soil type, and homogeneous land use. As a rule of thumb, catchment areas in undeveloped basins were targeted at 75 acres, but the size of individual catchments ranged from 30 - 300 acres. The target size for catchment areas in developed watersheds was also 75 acres, but catchment sizes varied from 50 to 300 acres. Overland flow lengths in undeveloped catchments were computed in the traditional way by area weighting typical overland flow lengths measured within the catchment. Depending on the catchment geometry, catchments drained either directly to a channel in the main stream system, or to a minor drainage way tributary to the main stream system.

For urbanized catchments, the overland flow length was taken as the average length from the back of a residential lot to the middle of the street serving the residence, which results in a very wide catchment. To account for gutter and pipe routing within the catchment, the watershed was drained to a routing channel whose length was determined by following the drainage system from its outfall to the centroid of the catchment, and whose cross-section was taken as the cross section of the catchment outfall, which could be either a pipe or an open channel.

Infiltration was simulated using the Horton Infiltration equation. The Horton model takes four input parameters: the maximum infiltration capacity of the soil, the minimum or final infiltration capacity of the soil, a decay coefficient which controls how quickly the soil loses its infiltration capacity, and the total infiltration that has occurred from the beginning of the storm. While the Horton model is the most widely-known model for infiltration, it is dependent on discrete field measurements or limited parameter values from the literature. In this study initial estimates of the Horton infiltration values for maximum and final infiltrations were assigned from area weighted values of SCS hydrologic soil groups from GIS data (USNRCS, 1994). Minimum infiltration values (f_c) were estimated by hydrologic soil groups (Huber and Dickenson, 1998) and maximum infiltration values (f_o) were linked to watershed soils through soils textures (sand, loam, silt, clay) described in the SWMM5 model help menu (U.S. EPA, 2006). These infiltration values are listed in Table 4-6. These values were not field tested in the individual watersheds, but rather were used as calibration variables to match model simulations of flow-duration curves and peak-flow frequency curves in each watershed to measured in-stream values (see Section 4.2.3 below). Final infiltration parameter values are given in Table 4-7. Infiltration rate decay constants (K) and times for a fully saturated soil to dry completely (T_w) were also based upon U.S. EPA (2006) recommendations from the SWMM 5 model help menu, and are shown in Table 4-7. It should be noted that minimum infiltration values (f_c) for several of the more urbanized watersheds (Black, Beaverdam, Pigeon, and Dutchmans) were higher than the values recommended by those described in the SWMM5 model help menu (U.S. EPA, 2006). These high values reduced peak runoff values of the largest storms, which was necessary to match peak

Table 4-6. Horton Infiltration Values.

Hydrologic Group	f_o (in/hr)	f_c Ranges (in/hr) High	f_c Ranges (in/hr) Low	Average f_c (in/hr)
A (sand)	7.5	0.45	0.30	0.375
B (loam)	4.5	0.30	0.15	0.225
C (silt)	3.0	0.15	0.05	0.100
D (clay)	1.5	0.05	0.00	0.025

Source: U.S. EPA (2006).

Table 4-7. Watershed Parameters Used in SWMM Models.

Watershed	Imperviousness				Horton Infiltration				Depression Storage		Mannings Overland Roughenss	
	GIA	DCIA	K_1	K_2	f_o (in/hr)	f_c (in/hr)	T_w (days)	K (1/hr)	Imperv.	Perv.	Imperv.	Perv.
Morgan	N.A.	0.8	N.A.	N.A.	4.50	0.40	7.0	4.00	0.05	0.20	0.010	0.30
Strouds	N.A.	3.8	N.A.	N.A.	4.50	0.40	7.0	4.00	0.05	0.20	0.010	0.30
Camp	N.A.	2.9	N.A.	N.A.	4.50	0.50	7.0	4.00	0.07	0.25	0.012	0.30
Wilson	9.0	9.0	1.000	1.000	4.47	0.45	7.0	4.14	0.07	0.26	0.012	0.26
Dutchmans	N.A.	4.4	N.A.	N.A.	4.50	0.60	7.0	4.00	0.07	0.25	0.012	0.30
Black	20.5	8.2	0.136	1.350	4.42	0.60	7.0	2.00	0.07	0.25	0.012	0.30
Beaverdam	11.7	4.9	0.145	1.405	4.21	0.70	7.0	1.50	0.07	0.15	0.012	0.26
Pigeon	30.6	14.3	0.130	1.350	4.23	0.70	7.0	1.50	0.07	0.25	0.012	0.30

N.A. = Not applicable because DCIA was calculated directly from road areas.

flow frequency exceedance and flow duration curves. The model was minimally sensitive to maximum infiltration (f_o) values, nor was it sensitive to times for a fully saturated soil to dry completely (T_w). The model was more sensitive to changes in infiltration rate decay constants (K), and this value was used for model calibration, as well. Extensive discussion of model parameterization and calibration can be found in Davis (*in preparation*).

The level of imperviousness, or gross imperious area (GIA) of each subarea in the highly urbanized watersheds (Black, Pigeon, and Beaverdam), was computed from the 2001 National Land Cover Database Zone 60 Imperviousness Layer (USGS, 2003). In this study, directly connected impervious area (DCIA) was modeled, rather than GIA, in order to more accurately represent the effect of runoff from roofs and parking lots that are routed to pervious surfaces. The DCIA for the urban catchments was estimated using the following equation (Lee and Heaney, 2003).

$$DCIA = K_1 \cdot GIA^{K_2} \tag{4-1}$$

The original parameters of K_1 and K_2 proposed by Lee and Heaney (2003), 0.15 and 1.41, respectively, were modified for the North Carolina area during the calibration process by comparing simulated peak-flow frequency exceedence curves to peak-flow frequency exceedence curves computed from the level-flow monitoring data collected by the USGS from 2002-2003. In the study watersheds the K_1 and K_2 values ranged between 0.13 and 0.145 for K_1 and 1.35 and 1.405 for K_2. The DCIA in the rural watersheds (Morgan, Strouds, Camp, Dutchmans, and Wilson) was estimated from the area of the roads in each subcatchment, as measured from aerial photos (USGS, 2006b). DCIA and GIA values, as well as K_1 and K_2 values are shown in Table 4-7.

Stream channels were modeled as irregularly-shaped cross sections using data collected by CSU staff during the November 2005 field surveys (see Section 4.2.1). Floodplains were estimated using 2-foot contour data (Wake County, 1999; NCDOT GIS, 2005) to allow for the inclusion of overbank flow in the model without creating a "flooding" condition within the model. The bottom slope of each stream channel segment modeled was also measured from the 2-foot contour maps and channel lengths were calculated using ArcMap GIS.

Fifteen-minute precipitation data from USGS Site No 0208732885 (Marsh Creek near New Hope, NC) were applied to each of the models. A twenty year period of record was modeled, from January 1, 1985 to December 31, 2004.

Interflow and groundwater-streamflow interactions were found to be important for matching the portion of the flow-duration curve representing the recession limb of runoff hydrographs. Interflow was accounted for in the models by applying the groundwater simulation algorithms in SWMM5 to each of the watersheds. In most watersheds, a single aquifer was modeled for each watershed, with the aquifer accepting surface water infiltration and diverting it to one receiving node near the outlet of the watershed. The Morgan Creek watershed was modeled with three aquifers with watersheds from the two headwater tributaries contributing groundwater to two receiving nodes near the outlets of the respective tributaries, and a third aquifer contributing infiltration from the rest of the watershed to a receiving node near the outlet of the watershed. Groundwater parameters were used to calibrate the models to the recession limb flow conditions.

A summary of groundwater parameters used for each watershed is given in Table 4-8, and groundwater equation variables are given in Table 4-9. In general, aquifers were modeled as of sandy clay, and parameters from Rossman (2005) were applied. Hydraulic conductivity and the wilting point were used as calibration parameters to allow for a better match of the full flow duration curve, specifically aiding in the match of the recession limb of storm hydrographs. Hydraulic conductivity values greater than one are indicative of a high level of groundwater outflow that would be expected from a loamy sand or sandy aquifer.

Table 4-8. Groundwater Parameters Used in SWMM5 Models.

Watershed	Porosity (fraction)	Wilting Point (fraction)	Field Capacity (fraction)	Hydraulic Conductivity (in/hr)	Lower GW Loss Rate (in/hr)	Unsat. Zone Moisture
Morgan Creek	0.430	0.100	0.321	2.00	0.000	0.3
Strouds Creek	0.430	0.100	0.321	2.00	0.000	0.3
Camp Branch	0.453	0.085	0.321	3.00	0.000	0.3
Wilson Creek	0.430	0.100	0.321	5.00	0.000	0.3
Dutchmans Branch	0.430	0.221	0.321	0.02	0.002	0.3
Black Creek	0.430	0.221	0.321	0.02	0.0026	0.3
Beaverdam Creek	0.430	0.221	0.321	0.02	0.002	0.3
Pigeon House Creek	0.430	0.221	0.321	0.02	0.002	0.3

Table 4-9. Groundwater Equation Variables Used in SWMM5 Models.

Watershed	GW Flow Coeff.	GW Flow Exp.	SW Coeff.	SW Exp.
Morgan Creek	0.001	5	0.001	0
Strouds Creek	0.001	5	0.001	0
Camp Branch	0.010	5	0.000	0
Wilson Creek	0.010	7	0.001	1
Dutchmans Branch	0.010	1	0.0001	1
Black Creek	0.010	1	0.001	1
Beaverdam Creek	0.010	1	0.001	1
Pigeon House Creek	0.015	1	0.010	1

Groundwater model parameters were used to calibrate the models to match mid-range flows on the flow duration curve; the input parameters were not physically validated. Information regarding groundwater aquifer materials and elevations were not available for this study. Input parameters were based on literature values of material types, which are described in Table 4-10. Extensive discussion of model parameterization and calibration, as well as sensitivity of the model to groundwater input parameters can be found in Davis (*in preparation*). It is recognized that lack of supporting groundwater data is a source of uncertainty in the model results, however calibration of the peak flow

Table 4-10. Recommended Aquifer Parameter Values.

Soil Texture Class	Hydraulic Conductivity (in/hr)	Porosity (fraction)	Field Capacity (fraction)	Wilting Point (fraction)
Sand	4.74	0.437	0.062	0.024
Loamy Sand	1.18	0.437	0.105	0.047
Sandy Loam	0.43	0.453	0.19	0.085
Loam	0.13	0.463	0.232	0.116
Silt Loam	0.26	0.501	0.284	0.135
Sandy Clay Loam	0.06	0.398	0.244	0.136
Clay Loam	0.04	0.464	0.310	0.187
Silty Clay Loam	0.04	0.471	0.342	0.21
Sandy Clay	0.02	0.430	0.321	0.221
Silty Clay	0.02	0.479	0.371	0.251
Clay	0.01	0.475	0.378	0.635

Source: Rossman, 2005.

frequency exceedance and flow duration curves in the discharge ranges that affect the hydrologic metrics considered in this study provides a level of confidence that model uncertainty is minimized, although not eliminated.

4.2.3 Model Calibration

Calibration of seven of the eight watersheds was conducted by comparing model output to the level-flow monitoring data collected by the USGS from 2002-2003. The model for the Strouds Creek watershed was not calibrated because flow data were not provided by the USGS for this watershed. For the Strouds Creek watershed model, input parameters that would typically be calibrated were set equal to final parameters determined by calibration for the Morgan Creek watershed. The Morgan Creek watershed is the watershed included in this study that is located closest to the Strouds Creek watershed; it has similar simplified USDA soil texture classifications and hydrologic soil groupings to those in the Strouds Creek watershed.

Calibration techniques differed depending on if the watersheds were urban or rural. Urban watersheds (Pigeon, Black, and Beaverdam) were calibrated mainly with adjustments in DCIA, while the rural watersheds (Morgan, Camp, Dutchmans, and Wilson) were calibrated with adjustments in infiltration. As stated in the previous section, groundwater parameters were used to calibrate the models to flow conditions that occurred between rainfall events.

Traditionally, stormwater model calibration is performed on six or more individual storms of varying size (Akan and Houghtalen, 2003); it is considered complete for the single storm event when the peak discharge and volume of runoff are accurately reflected by the model. Due to the importance of accurately simulating the full spectrum of flows for the computation of hydrologic and geomorphic metrics necessary to create linkages between land use patterns and runoff control practices to stream biologic integrity, the models were calibrated across the full spectrum of flows that occurred during the 2002-2003 period of level-flow monitoring by the USGS.

Model parameters were adjusted to match flow-duration and peak-flow frequency exceedence curves for 15-minute model output to 15-minute level-flow data measured by the USGS at the watershed outlets. Any five-minute data collected by the USGS during the 2002-2003 level-flow monitoring were converted to 15-minute data for the comparison. The flow duration curves were the primary tool for calibration in the mid-to low-flow range and peak flow frequency exceedence curves were used for calibration of the magnitude of runoff event peaks. The flow-duration curve is a graph of all the discharges during a continuous record, and their cumulative exceedences, or the percent of time each discharge occurs during the period of record. Figure 4-3 shows the flow

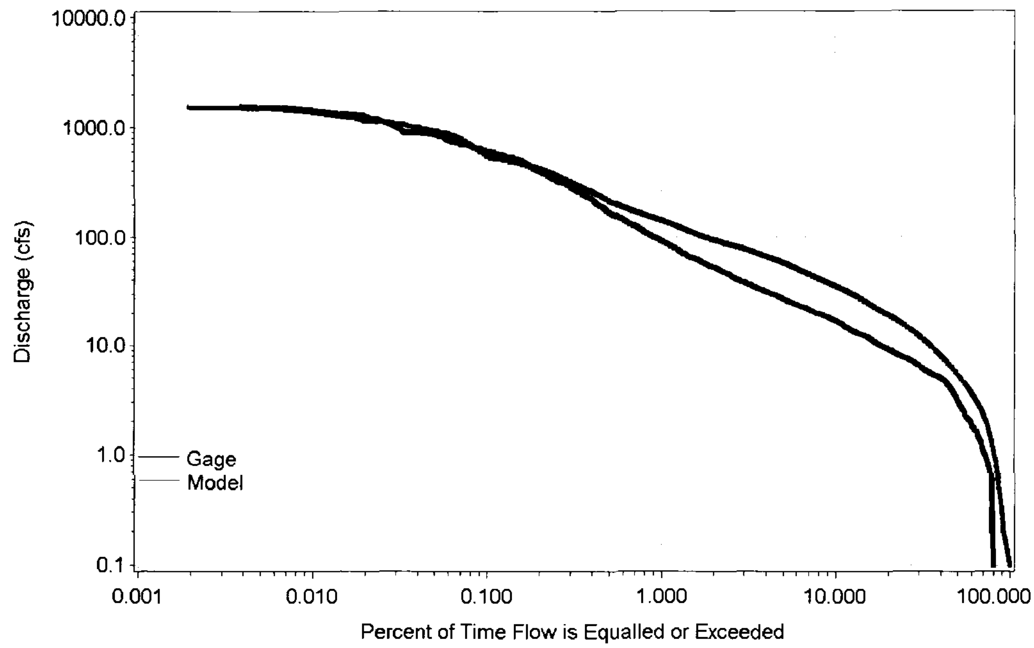


Figure 4-3. Morgan Creek Watershed Flow Duration Calibration Period.

duration results from the calibration of the Morgan watershed. Flow duration curves for the period of calibration for the six remaining watersheds are provided in the Appendix in Figures A-1, A-3, A-5, A-7, A-9 and A-11.

Peak flow frequency exceedence curves were developed from the partial duration series of peak flows. This approach, which is in contrast to the examination of the annual maximum series, was used because it allowed for the analysis of high frequency, low runoff producing storms. In addition, this approach describes the true frequency with which a given flow occurs as opposed to simply describing the annual maximum flow. This is important because erosion potential and biotic response of instream fauna is determined by how often a given flow is equaled or exceeded, not just the annual high flow.

Calculation of the partial duration series requires definition of an inter-event time, and the minimum threshold to define an event. A six-hour inter-event time was specified to separate the flow data into individual events based on recommendations by Heineman (2007). A sensitivity analysis was performed on the gage flow data from each watershed to identify the minimum threshold value to use to define an event. Table 4-11 shows that the Pigeon House Creek, Beaverdam Creek, Black Creek and Camp Creek watersheds were not highly sensitive to the minimum threshold value to define an event, but the other watersheds (Morgan Creek, Wilson Creek, Dutchmans Creek) were not found to be more sensitive to this parameter. Sensitivity analysis was not conducted for the Strouds Creek watershed because flow data were not provided by the USGS for this site. Final event thresholds parameters for each watershed were ultimately selected to identify the threshold value which gave the greatest number of events during the period of USGS flow monitoring. The event thresholds used for each watershed are given in Table 4-11. Figure 4-4 shows the peak flow frequency exceedence results from the calibration of the Morgan Creek watershed. Peak flow frequency exceedence curves for the period of

Table 4-11. $Q_{0.5}$ Values Computed Using Varying Event Thresholds.

Watershed	Variable		Event Thresholds 0.5 cfs	1 cfs	2 cfs	3 cfs	4 cfs	5 cfs	6 cfs	7 cfs	8 cfs	9 cfs	10 cfs
Pigeon	(3.0)	758	378	428	758	758	758	758	758	758	758	758	758
Beaverdam	(2.0)	392	235	392	392	392	392	392	392	392	392	392	392
Morgan	(10.0)	384	1	38	70	92	92	165	165	165	239	384	384
Wilson	(4.5)	128	42	87	121	128	128	128	128	248	248	248	248
Black	(0.9)	495	495	495	495	495	495	495	495	495	495	495	495
Dutch	(10.0)	877	1	4	4	4	*	*	**	877	877	877	877
Camp	(2.0)	135	13	135	135	135	135	135	135	135	135	135	135

*Only 3 events identified using this threshold, therefore a 0.5-year storm not identified

**Only 2 events identified using this threshold, therefore a 0.5-year storm not identified

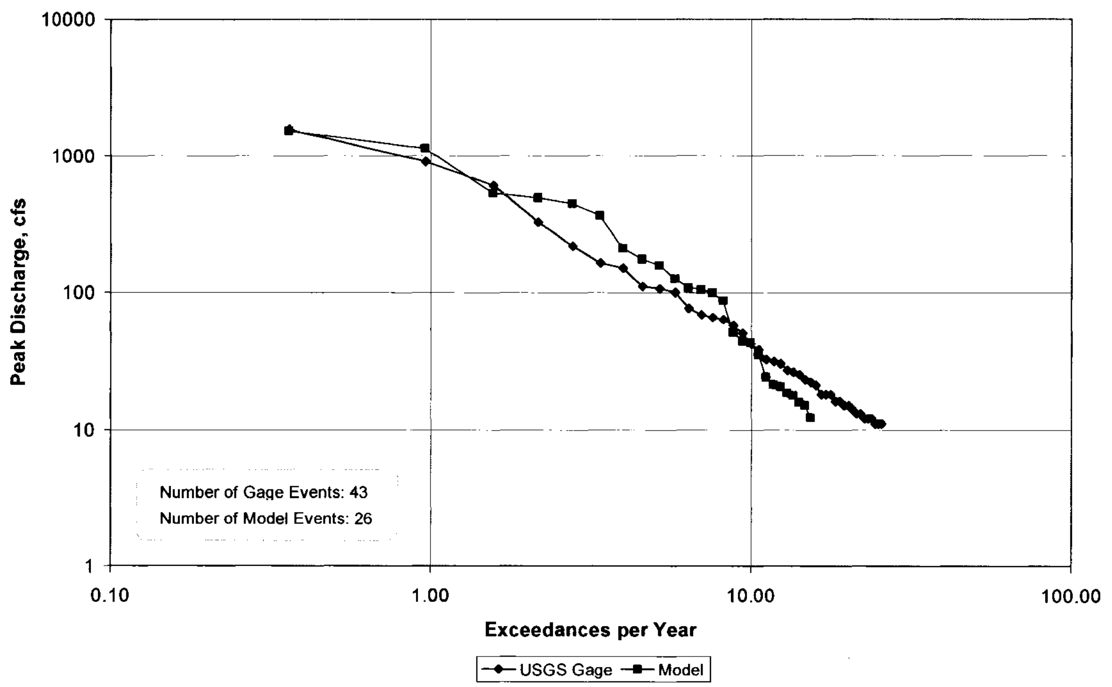


Figure 4-4. Morgan Watershed Peak Flow Frequency Exceedance Calibration.

calibration for the six remaining watersheds are given in the Appendix in Figures A-2, A-4, A-6, A-8, A-10 and A-12.

The Cunnane (1978) formula was used to calculate the frequency of an event peak flow, *T*, as:

$$T = \frac{N+1-2A}{M-A} \tag{4-2}$$

where, T = return interval (years), N = number of years of record, M = rank of the event (in descending order of magnitude), and A = plotting position parameter (generally assigned a value of 0.4). The return interval was converted to exceedences per year, E, using:

$$E = \frac{1}{T}. \tag{4-3}$$

It should be noted that calibration of the SWMM models was satisfactory for all of the watersheds, except the Wilson Creek watershed, in the range of flows used to compute the $T_{0.5}$, T_1, and T_2 hydrologic metrics described in Section 4.2.5. Calibration to the peak flow frequency exceedance curve in the Wilson Creek watershed was poor, although calibration to the flow duration curve was satisfactory. Investigation of these results showed that the flow duration curve was matched because multiple events with short durations of high discharges were produced by the model, where the gage record showed longer durations of high flows for fewer individual events. Initially, this discrepancy was thought to have been caused by the distance between the rain gage used for modeling and the Wilson Creek watershed, however this phenomenon was not observed in the Morgan Creek watershed, which was equally distant (or farther) from the rain gage used for modeling. Examination of $T_{0.5}$ metric value computed from the Wilson Creek gage record showed that the $T_{0.5}$ value was out of the expected range (see Section 5.1), indicating that the gage record may be inaccurate. Further investigation of the gage data is warranted.

4.2.4 Modeling the Impact of Land Use and Runoff Control Strategies

The effects of land use and channel geometry on hydrologic and geomorphic metrics were evaluated in the Morgan Creek watershed. Alternative scenarios were evaluated; they involve investigation of the impacts of conversion of land use from rural (0.8 % DCIA) to urbanized (14.8 % DCIA), as well as the impacts of stream channel geometry on the metrics. Further discussion of these scenarios is given in Section 5.3.

Stormwater best management practices (BMPs) for flood control and water quality treatment were applied to the urbanized scenarios (14.8 % DCIA) in the Morgan

Creek watershed. Twenty-seven control points were modeled (Figure 4-5) to accomplish control of the water quality capture volume (WQCV) and peak shaving of the 2-yr and 10-yr design storms to the peak discharge values observed for these storms under rural watershed conditions (0.8 % DCIA). The WQCV was computed as the product of the mean runoff event and a drawdown coefficient, resulting in the capture of 70 - 90 percent of all runoff-producing events in their entirety. Using procedures recommended by WEF and ASCE (1998) and a draw-down time of 40 hours, the resulting WQCV was calculated as:

$$P_o = (ac)P_6 \tag{4-4}$$

where, a = drawdown coefficient (1.299), P_6 = mean storm precipitation over the watershed equal to18.0 mm (WEF and ASCE, 1998), and c = runoff coefficient. The runoff coefficient was calculated using:

$$c = 0.858i^3 - 0.78i^2 + 0.774i + 0.04 \tag{4-5}$$

where i = watershed imperviousness ratio (TIA). Orifice outlets for the 2-year and 10-year peak shaving were sized using an iterative procedure in Excel. All detention basins are 11-ft deep or less and have orifices varying in size from 0.5-ft to 2.5-ft in diameter. Basins were modeled with square foot-prints for simplicity. Two general outlet designs are illustrated in Figure 4-6. The outlet configuration used for each control point was variable because of differences in drainage area size, configurations required to allow draw-down the WQCV in 40 hours, and the need to vary orifice sizes and numbers to allow for matching of peak discharge values.

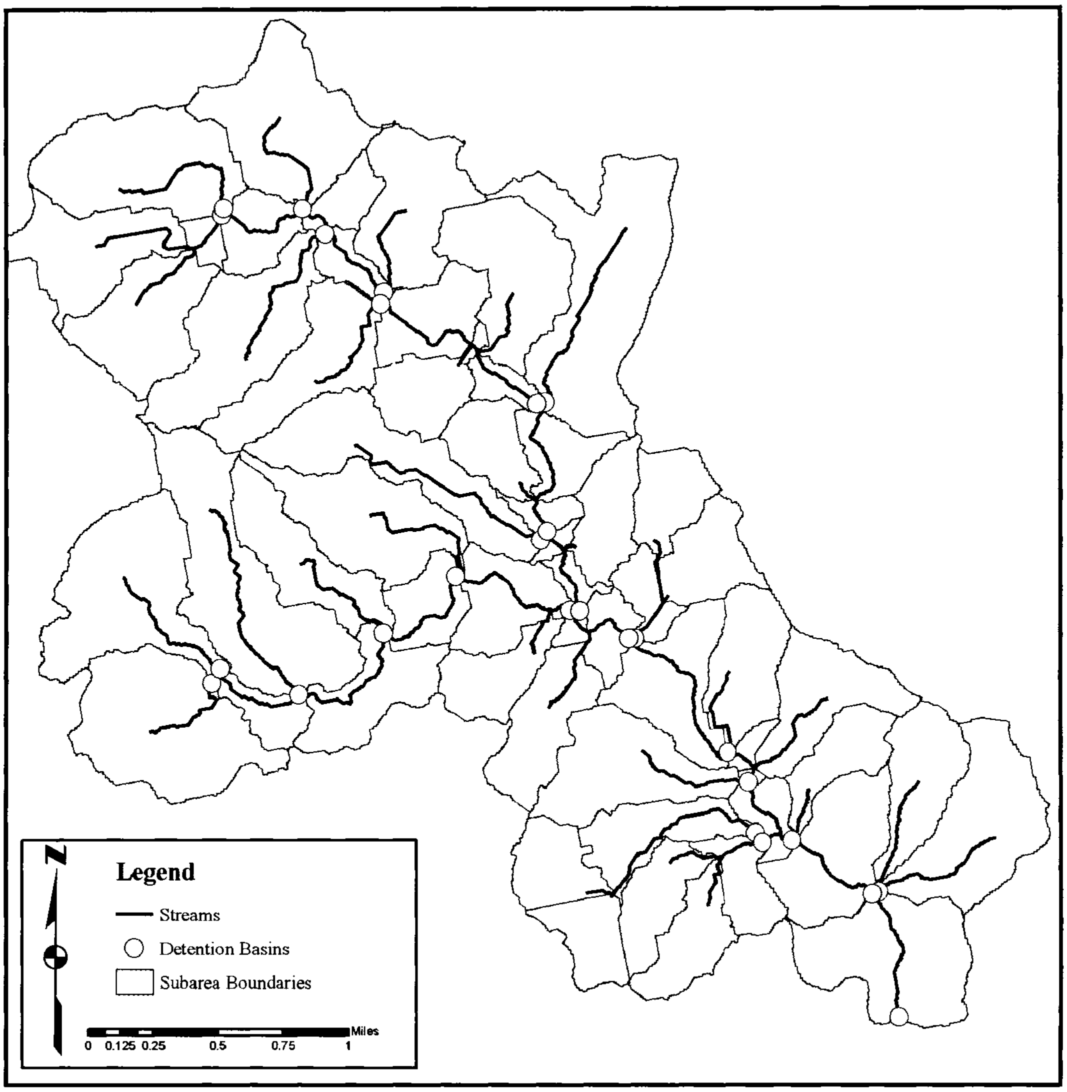


Figure 4-5. Location of Detention Basins Modeled in Morgan Creek Watershed.

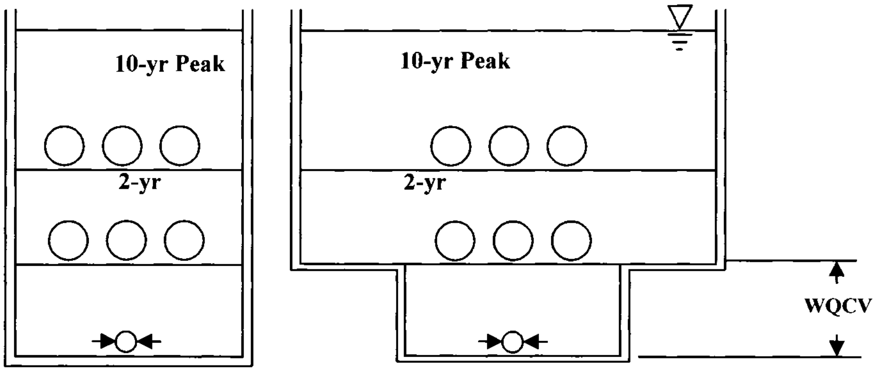


Figure 4-6. General Design of Stormwater Detention Pond Outlets.

4.2.5 Computation of Hydrologic and Geomorphic Metrics

Twenty years of precipitation data were applied to the SWMM5 models to generate continuous flow hydrographs at the outlets of the watersheds. These continuous flow records were used to compute hydrologic and geomorphic metrics for each of the watersheds. Four hydrologic metrics were computed using SAS™ from discharges generated by the SWMM5 models at 15-minute time intervals. The following hydrologic metrics were examined:

- The $T_{0.5}$ metric, modeled after Booth et. al. (2004), is a measure of stream flashiness, and in this analysis was evaluated as the percent of time flow is above the peak discharge of a half-year return interval storm. The half-year return interval storm was computed from a partial duration series.
- The T_1 metric, evaluated as the percent of time flow is above the peak discharge of a one-year return interval storm. The one-year return interval storm was computed from a partial duration series.
- The T_2 metric, evaluated as the percent of time flow is above the peak discharge of a two-year return interval storm. T_2 metrics were evaluated for two-year return interval storms computed from both a partial duration and an annual maximum series. The partial duration series has been shown to be sensitive to minimum flow thresholds for certain watersheds in this study (see Section 4.2.3), and this comparison allowed for examination of the impact of this to relationships between the T_2 and macroinvertebrate metrics.

- The T_{Qmean}, modeled after Booth et. al. (2004), which is the percent of time that the flow is above the mean annual flow.

The $T_{0.5}$ hydrologic metric was evaluated for six time periods:

- the 12-17 month periods of record surrounding the biologic data collection, which was the period of level/flow monitoring at each USGS EUSE site, and the period of record the models were calibrated to; approximately July, 2002 to November, 2003;
- the one-year period of record prior to the biologic data collection: July, 2002 to June, 2003;
- the two-year record prior to the biologic data collection: July, 2001 to June, 2003;
- the five-year record prior to the biologic data collection (July, 1998 to June, 2003);
- the 10-year record prior to the biologic data collection (July, 1993 to June, 2003); and
- the 20-year record (January, 1986 – December, 2005).

The T_1 metric was evaluated for the five-, ten-, and 20-year time periods described above. The T_2 and T_{Qmean} metrics were evaluated for slightly different time periods, to allow for easier computation of the annual maximum series values and mean annual flows for each calendar year. The following time periods were examined for the T_2 and T_{Qmean} metrics:

- the five-year record prior to the biologic data collection (1998-2002);
- the 10-year record prior to the biologic data collection (1993-2002); and
- the 20-year record prior to the biologic data collection (1986-2005).

Geomorphic metrics describing cumulative excess shear stress and erosion potential were calculated using SAS™ and water depth output from the SWMM5 models at 15-minute time intervals; the methods described in Section 3.4.2.4 were used for these computations. The geomorphic metrics were evaluated for the one-, two-, five-, 10-, and 20-year periods of record described for the $T_{0.5}$ metric above.

5.0 RESULTS FROM PROTOCOL TESTING IN THE NORTH CAROLINA PIEDMONT

5.1 Relationships Between the $T_{0.5}$ Hydrologic Metric and Watershed Characteristics

$T_{0.5}$ hydrologic metric results computed from the model were compared with those computed from the flow monitoring data provided by the USGS. Figure 5-1 shows that the metric values, while not identical, are linearly related. Results are shown for six of the sites. A $T_{0.5}$ value was not computed for the Strouds Creek watershed from monitoring data, as USGS discharge data were not available. It should be noted that the values in Figure 5-1 are not calibration results, but rather a comparison of the $T_{0.5}$

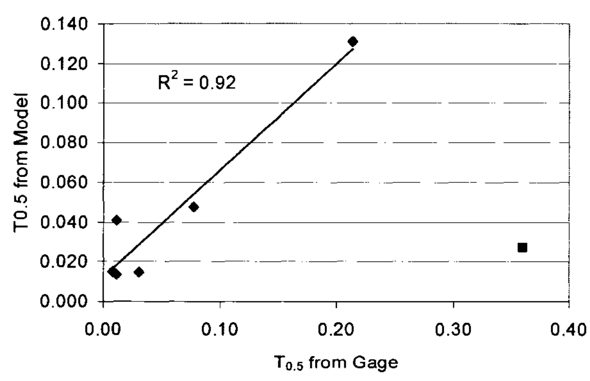


Figure 5-1. Comparison of Values of $T_{0.5}$ Hydrologic Metric from Gage and from Model During Period of Monitoring/Calibration.

computed from model simulations where the model was calibrated against the measured peak flow frequency exceedance and flow duration curves.

The $T_{0.5}$ value from the monitoring data for the Wilson Creek watershed was excluded from the regression (shown as the red square in Figure 5-1), because the $T_{0.5}$ value was an outlier when compared to the values computed for the other sites. Calibration of the model to the peak flow frequency exceedance curve for this watershed was poor, as discussed in Section 4.2.3. However, it should be noted that the $T_{0.5}$ computed from the model falls within the range expected for a watershed with the level of development that exists within the Wilson Creek watershed, but the $T_{0.5}$ value from the gage data is much higher than the $T_{0.5}$ values for all of the other watersheds. This indicates that there is likely a problem with the flow data provided by the USGS for this watershed, requiring further investigation.

Data were examined to identify relationships between the $T_{0.5}$ hydrologic metric and the urban intensity index values calculated by the USGS for each watershed. Figure 5-2 shows regressions of the $T_{0.5}$ values to the urban intensity index (UII) computed by the USGS (McMahon and Cuffney, 2000; Coles et al., 2004; Tate et al., 2005). These figures show a logarithmic relationship between the $T_{0.5}$ and the UII for all six time periods, with the strongest relationship occurring during the 10- and 20-year periods. Data were also examined to determine if basin size was influential on $T_{0.5}$ hydrologic metric values. Figure 5-3 shows that when all eight watersheds are included in the analysis, it appears that the values of the $T_{0.5}$ metric are influenced by drainage area. However, it should be noted that the largest watersheds included in this study (Morgan and Strouds Creeks, both with drainage areas greater than 8 square miles) were also the

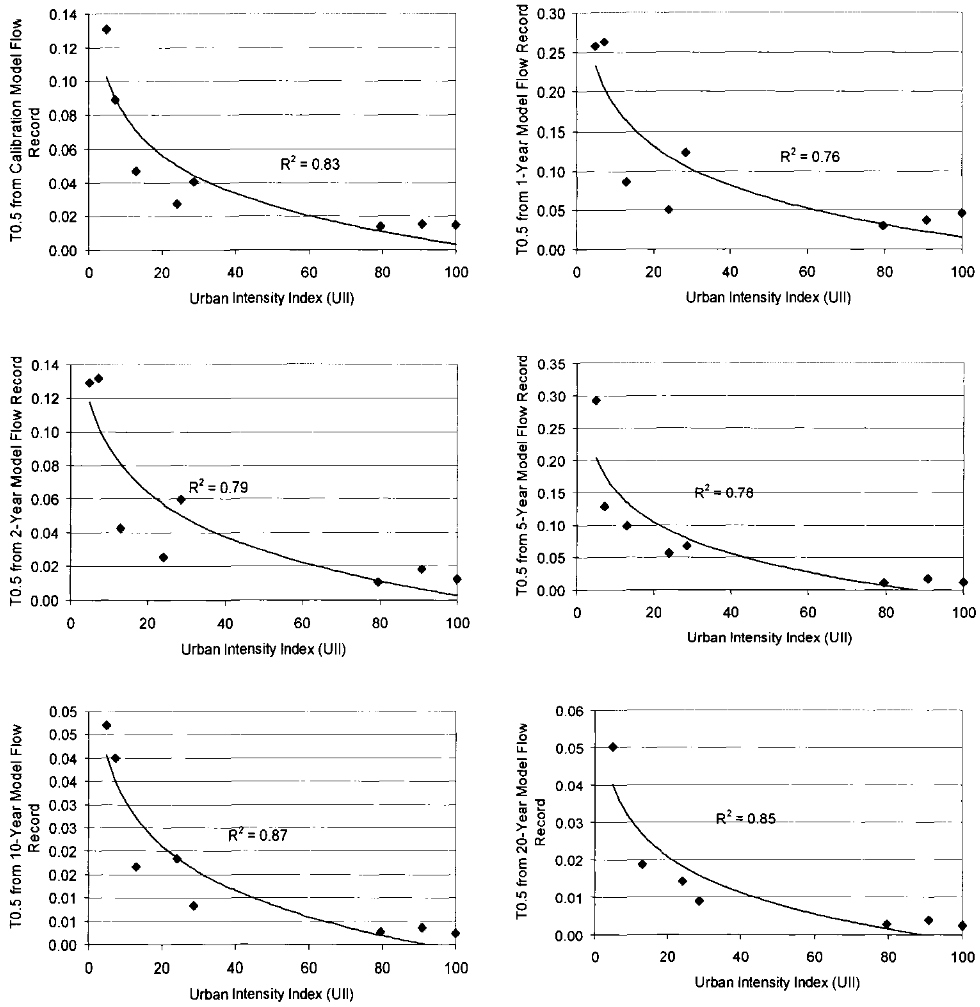


Figure 5-2. Relationship of the Urban Intensity Index to the $T_{0.5}$ Calculated From (a) the Calibration Period, (b) 1-Year Model Flow Record, (c) 2-Year Model Flow Record, (d) 5-Year Model Flow Record, (e) 10-Year Model Flow Record, and the (f) 20-Year Model Flow Record.

watersheds with the least amount of development, and were expected to have the highest $T_{0.5}$ values. When these two watersheds are excluded from the data set, no relationship between drainage area and the $T_{0.5}$ metric was observed, shown in Figure 5-3. When values of the $T_{0.5}$ metric were computed for various locations within the Morgan Creek watershed, no relationship between drainage area and the $T_{0.5}$ metric was observed, as is shown in Figure 5-4. This finding is consistent with that of Konrad et al. (2005), who showed that the $T_{0.5}$ was insensitive to basin size in Puget lowland stream basins.

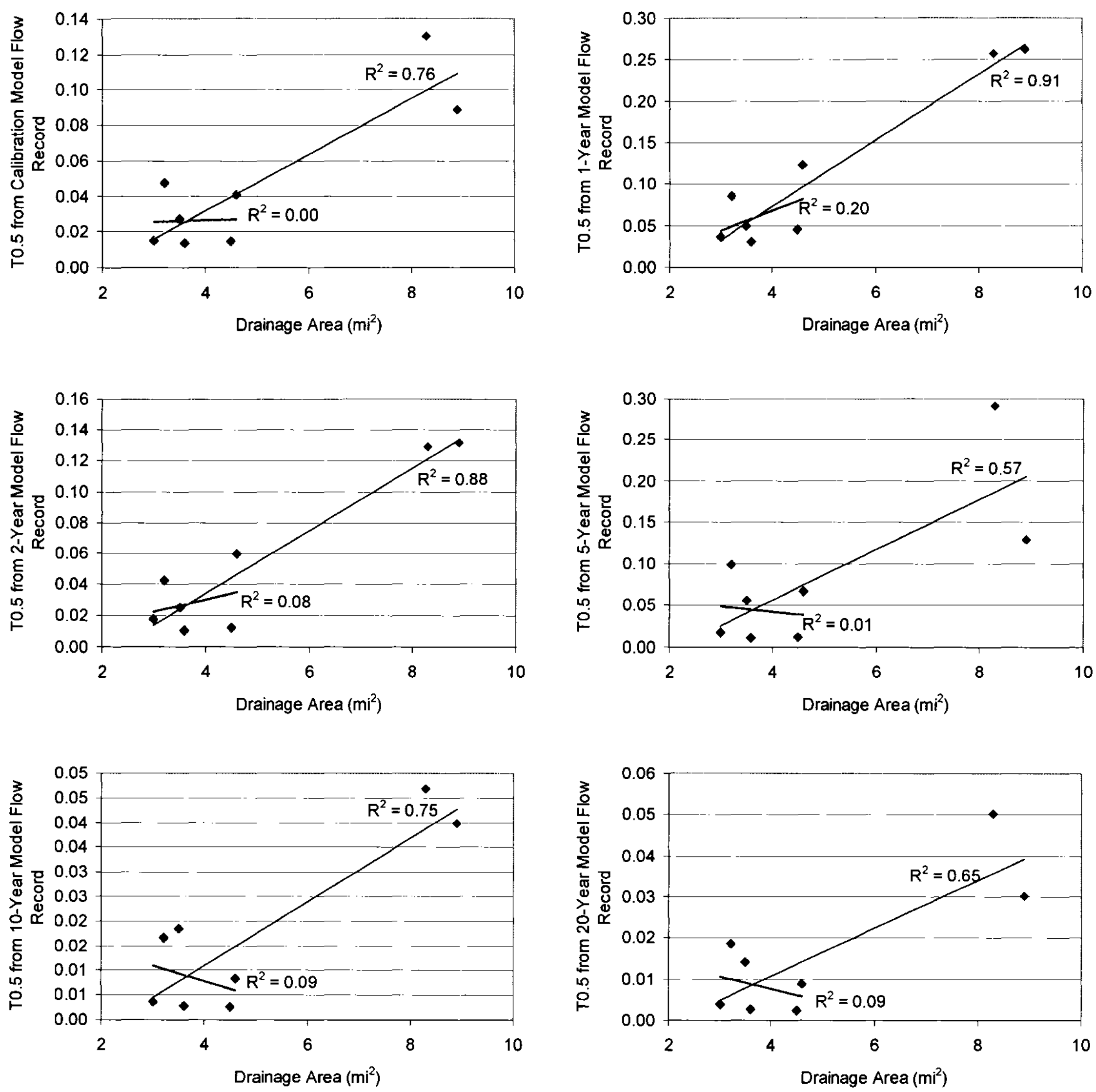


Figure 5-3. Influence of Drainage Area on the $T_{0.5}$ Calculated From (a) the Calibration Period, (b) 1-Year Model Flow Record, (c) 2-Year Model Flow Record, (d) 5-Year Model Flow Record, (e) 10-Year Model Flow Record, and (f) 20-Year Model Flow Record.

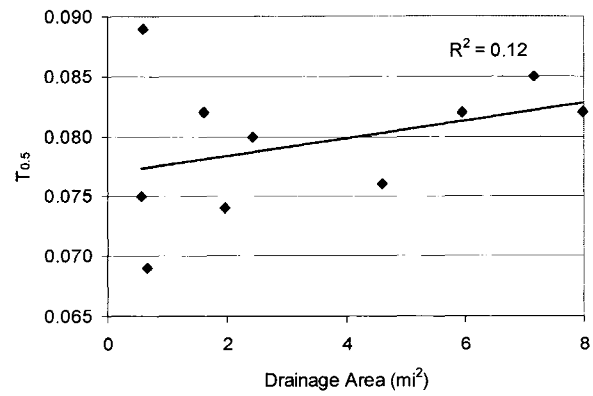


Figure 5-4. Influence of Drainage Area on the $T_{0.5}$ Calculated for Various Locations Within the Morgan Creek Watershed.

5.2 Establishing Linkages between Hydrologic and Geomorphic Metrics and Biologic Metrics

5.2.1 Relationships between Hydrologic and Macroinvertebrate Metrics

Step 5 of the protocol described in Section 3.7 was used to evaluate relationships between hydrologic metrics and macroinvertebrate metrics. Figure 5-5 shows the relationship between results of benthic macroinvertebrate sampling conducted by the USGS in the study watersheds and the $T_{0.5}$ values computed from the level/flow monitoring gages in each watershed. A positive linear response is observed; however, similar to the results shown in Figure 5-1 in the previous section, the results from the Wilson Creek watershed were excluded from the regression analysis. Note, again, that the $T_{0.5}$ value for the Morgan Creek watershed is well out of range of the values identified for the other watersheds, and is thought to be out of range because of bad monitoring data. Results for EPT richness, a count of the total EPT specimens collected for both richest

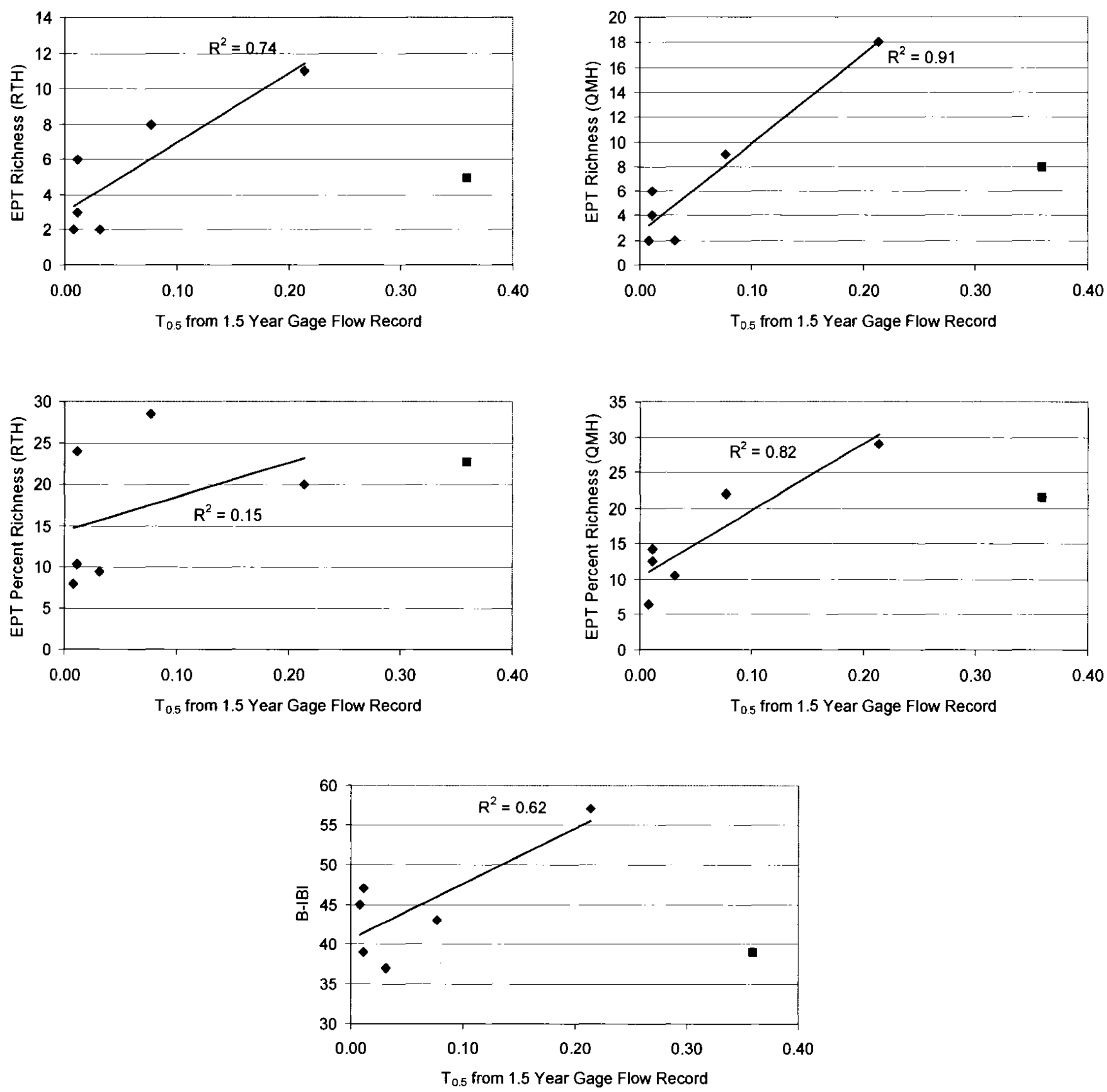


Figure 5-5. Relationship of $T_{0.5}$ Calculated From the 1.5-Year Gage Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

targeted (RTH) and qualitative multi-habitat (QMH) are displayed in addition to results for EPT percent richness, which is the percent of EPT taxa collected in relationship to the total number of taxa collected. Benthic index of biotic integrity (B-IBI) results are also regressed against $T_{0.5}$ values. The B-IBI values were computed using methods described by Kerans (1994) for rivers of the Tennessee Valley. A B-IBI method is not available specifically for the North Carolina Piedmont ecoregion.

Table 5-1 shows regression results for the $T_{0.5}$ hydrologic metric computed for both the gage data and the model output data. Table 5-1 shows that similar correlations were achieved from the $T_{0.5}$ values computed from the gage flow record and the flow record from the model for EPT richness, but correlations were much more different between EPT percent richness and B-IBI for the $T_{0.5}$ metric values of the gage and the model.

Table 5-1. Summary of Regression Results for the $T_{0.5}$ Hydrologic Metrics and Descriptors of Urbanization.

Macroinvertebrate Metric		$T_{0.5}$ (Model)							Land Use	
	$T_{0.5}$ (Gage)	Calibration Period	1-Year	2-Year	5-Year	10-Year	20-Year	Mean	UII	DCIA
EPT Richness (RTH)	0.74	0.74	0.86	0.88	0.56	0.82	0.69	0.76	0.71	0.57
EPT Richness (QMH)	**0.91**	**0.90**	0.87	0.89	**0.79**	**0.97**	**0.91**	**0.89**	0.72	0.53
EPT % Richness (RTH)	0.15	0.30	0.38	0.40	0.22	0.38	0.29	0.33	**0.84**	0.36
EPT % Richness (QMH)	0.82	0.72	0.69	0.71	0.64	0.87	0.77	0.73	0.80	0.37
B-IBI	0.62	0.78	**0.91**	**0.92**	0.61	0.73	0.67	0.77	0.42	**0.70**
Mean R^2	0.65	0.69	0.74	0.76	0.56	0.75	0.67	0.70	0.70	0.51

Figure 5-6 shows regressions of the $T_{0.5}$ hydrologic metric values (computed from modeling results for the two years prior to biologic sampling) to results of the benthic macroinvertebrate sampling conducted by the USGS in the study watersheds. Overall, relationships between the $T_{0.5}$ values and the macroinvertebrate data were strongest for $T_{0.5}$ values computed from the one-, two-, and 10-year flow records. R^2 values from the regression for each time period analyzed are summarized in Table 5-1 Regression plots for $T_{0.5}$ values from each time period analyzed are provided in the Appendix (Figures A-13 through A-17).

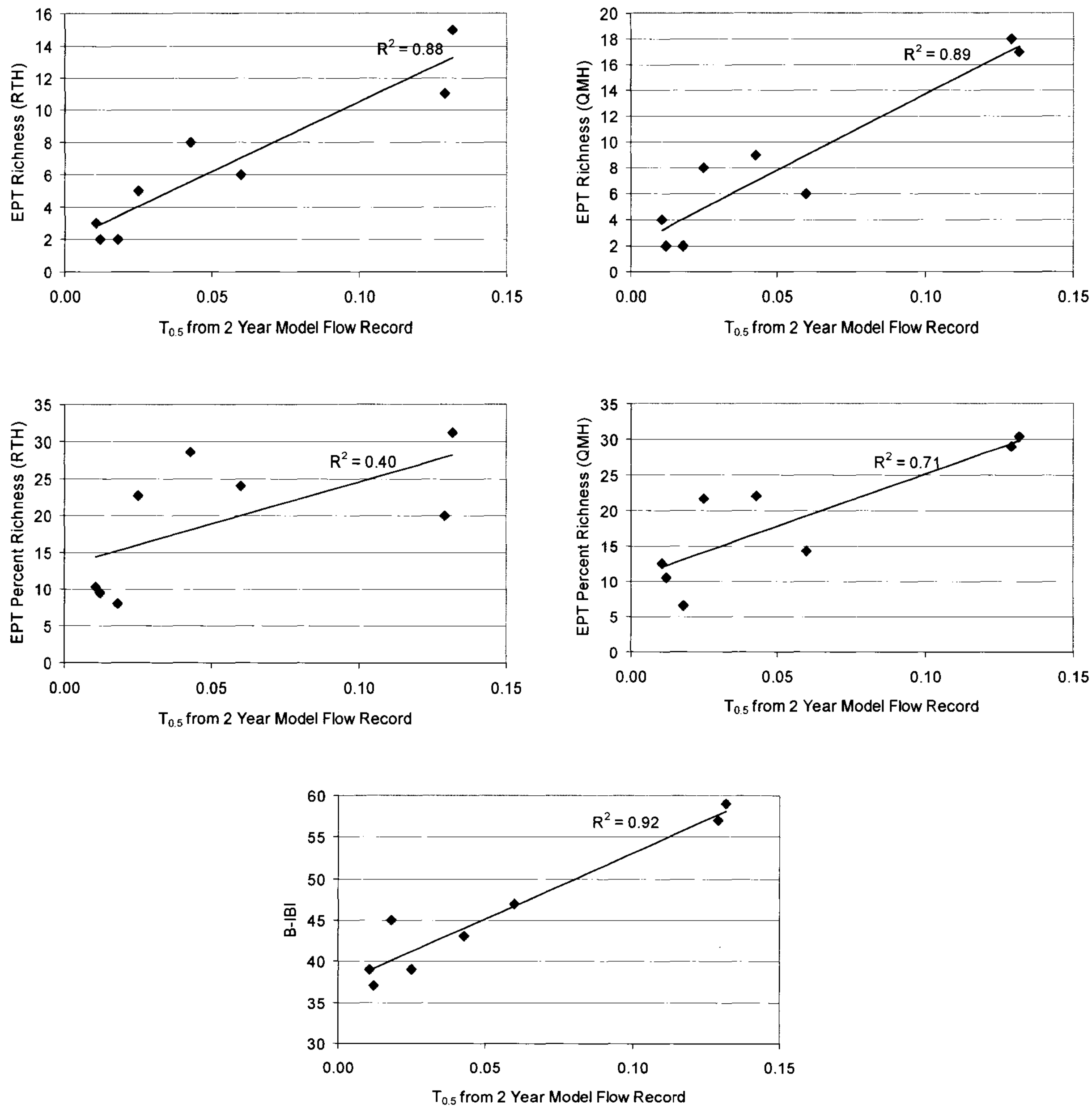


Figure 5-6. Relationship of $T_{0.5}$ Calculated From the 2-Year Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

The weaker correlations with $T_{0.5}$ values from the 5-year time period can possibly be explained by the fact that Hurricane Floyd occurred in 1999, which was within the 5-year time period prior to macroinvertebrate sampling. Hurricane Floyd was a category 4 (very strong) hurricane on the Saffir-Simpson Hurricane Scale (State Climate Office of North Carolina, 2007) that contributed 5.79 inches of rain in a 27-hour time period at the National Weather Service Raleigh Durham International Airport (Coop ID 317069) rain gage. Another possible explanation for weaker correlations during the 5- and 20-year time periods is that average annual precipitation values for these time periods were less than those for the 1-, 2- and 10-year time periods. Variations in precipitation affect the frequency, magnitude, and duration of flows in streams, which in-turn affect the $T_{0.5}$ hydrologic metrics. Further examination of the effects of precipitation variability on the $T_{0.5}$ hydrologic metric were not explored in this analysis, but may be warranted.

In general, the strongest relationships between biologic results and $T_{0.5}$ values were observed for EPT richness and for B-IBI. EPT percent richness (RTH) had the weakest correlations in all time periods examined. This weak correlation is attributed to the macroinvertebrate score for the Morgan Creek watershed, which appears to be an outlier with respect to corresponding $T_{0.5}$ values. Interestingly, this value does not appear to be an outlier when plotted against the UII (Figure 5-7).

Figure 5-7 shows a negative correlation between the macroinvertebrate metrics and the UII, with the strongest relationships occurring between the EPT percent richness metrics, and a low correlation between B-IBI and UII. Relationships between the macroinvertebrate metrics and directly connected impervious area (DCIA) were also examined, since imperviousness has historically been indicated as a predictor of aquatic community health (Center for Watershed Protection, 2002). Figure 5-8 shows that the EPT richness metrics correlate weakly with DCIA, however, there is a stronger relationship between DCIA and B-IBI.

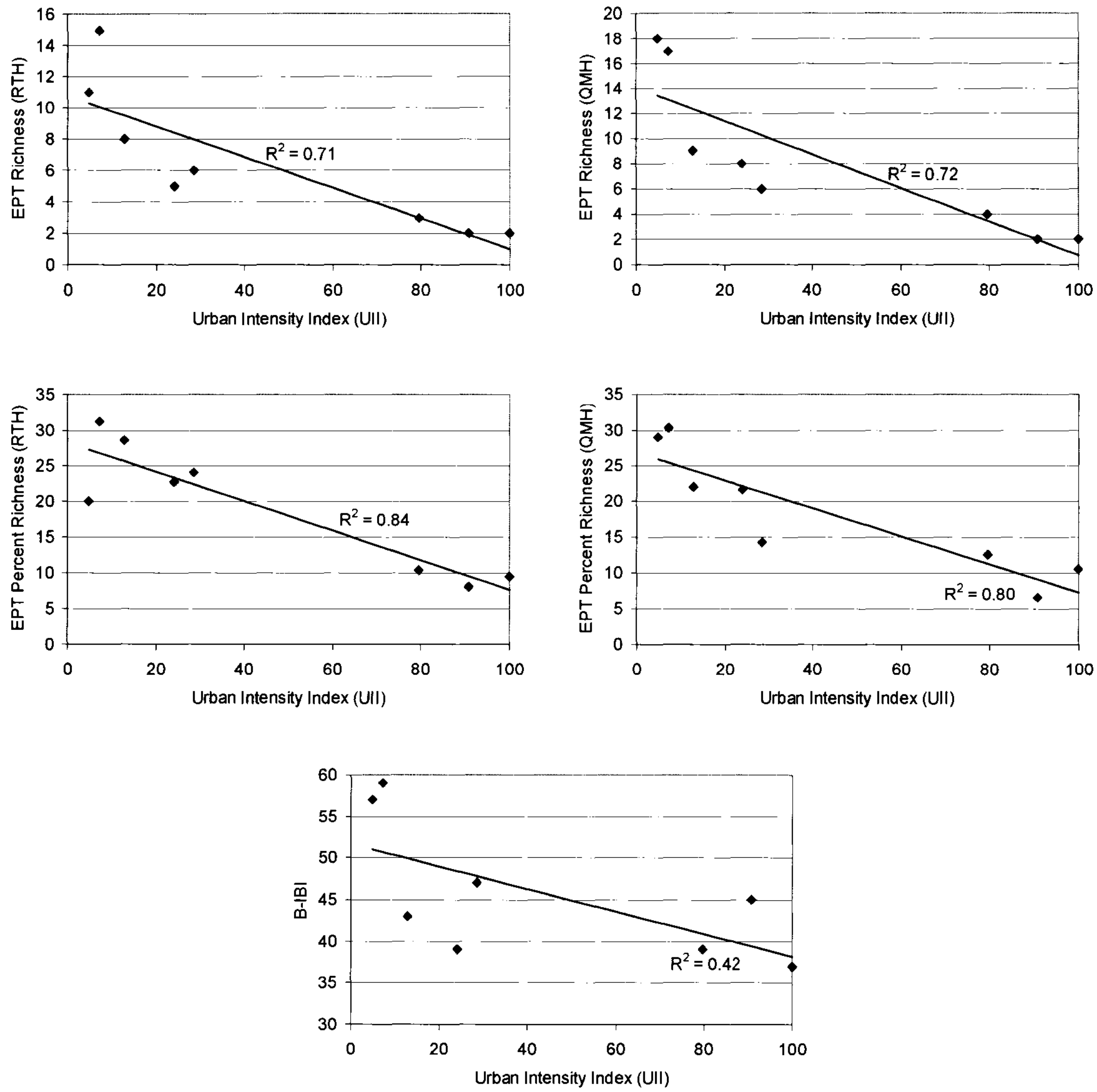


Figure 5-7. Relationship of the Urban Intensity Index to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

It is important to note that while relationships between macroinvertebrate metrics and measures of urbanization such as the UII and DCIA exist, these relationships offer no guidance on opportunities to prevent degradation of biotic integrity of streams, short of limiting the extent of urbanization in a watershed. Evaluation of the relationships between macroinvertebrate metrics and hydrologic metrics such as the $T_{0.5}$ allows

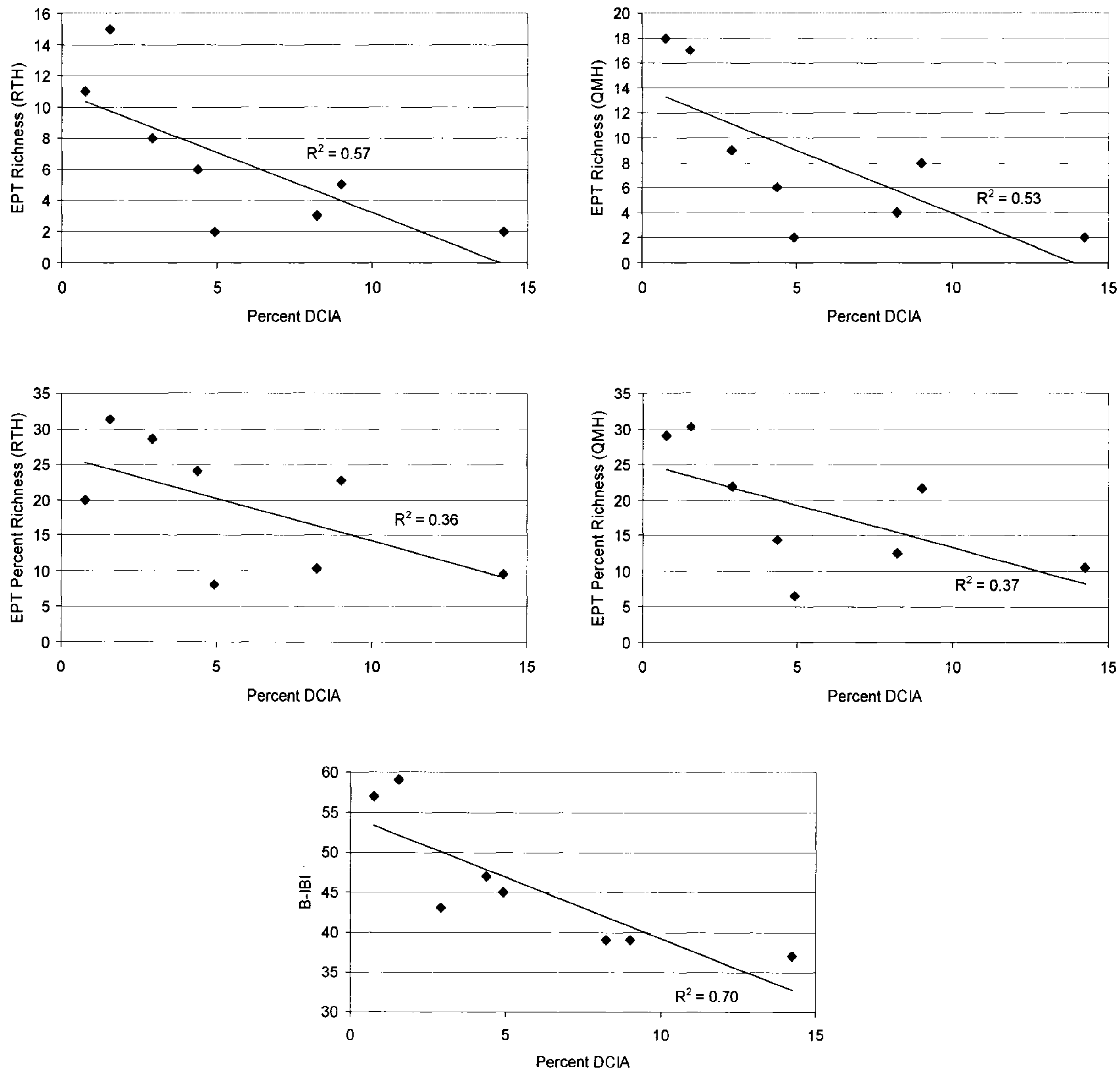


Figure 5-8. Relationship of DCIA to Benthic Macroinvertebrate Sampling Data (a) EPT (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

investigators to identify flow regimes that are conducive to the desired biotic outcome, and offers the opportunity to identify stormwater management practices that will produce the desired flow regime. However, the fact that not all measures of biologic health are strongly correlated to the $T_{0.5}$ means that stormwater management decision should not be based solely on analysis of the $T_{0.5}$, but should include knowledge gained from the complete set of data available. The effects of stormwater management practices on the $T_{0.5}$ metric in the North Carolina Piedmont is discussed in Section 5.3.1.

The $T_{0.5}$ hydrologic metric was also computed at hourly and daily time steps to determine the sensitivity of the metric to temporal resolution. Table 5-2 provides a summary of the regression results for the timestep comparison. It was found that the $T_{0.5}$ hydrologic metric computed from an hourly time step correlated with biologic data similarly to the $T_{0.5}$ hydrologic metric values computed from a 15-minute time step; however, the relationships between the macroinvertebrate metrics and the $T_{0.5}$ metric values were always stronger for $T_{0.5}$ values computed from the 15-minute data than values computed from the hourly data. The $T_{0.5}$ hydrologic metric computed from a daily timestep did not correlate well with the biologic data; for the 20 year flow record the range in the metric values computed for each watershed from the daily timestep was minimal (0.561 - 0.578).

Table 5-2. Summary of Regression Results from Timestep Comparison

Macroinvertebrate Metric	2-year			5-year			20-year			Average		
	15-min	hourly	daily	15-min	hourly	daily	15-min	hourly	daily	15-min	hourly	daily
EPT Rich (RTH)	0.88	0.81	0.35	0.56	0.53	0.02	0.69	0.72	0.02	0.71	0.69	0.13
EPT Rich (QMH)	0.89	0.85	0.30	**0.79**	**0.76**	0.04	**0.91**	**0.81**	0.04	**0.86**	**0.81**	0.13
EPT % Rich (RTH)	0.40	0.28	**0.63**	0.22	0.21	0.00	0.29	0.43	0.00	0.30	0.31	0.21
EPT % Rich (QMH)	0.71	0.64	0.50	0.64	0.61	**0.08**	0.77	0.76	**0.08**	0.71	0.67	**0.22**
B-IBI	**0.92**	**0.91**	0.04	0.61	0.58	0.00	0.67	0.55	0.00	0.73	0.68	0.01
Mean R^2	0.76	0.70	0.36	0.56	0.54	0.03	0.67	0.65	0.03	0.66	0.63	0.14

Results from the EPT richness (QMH) and the $T_{0.5}$ metrics regression analyses for the 10-year time period combined with the NCDENR (2006) EPT richness rating scores shown in Table 5-2 were used to identify $T_{0.5}$ values that would create flow conditions favorable to desirable EPT richness rating scores. An example of this process is illustrated in Figure 5-9. It is important to note that Figure 5-9 shows approximate ranges; the displayed values are not intended to be absolutes of thresholds and there is a band of uncertainty associated with the dashed lines that separate the "Poor", "Fair" and "Good-Fair" ranges.

The T_1 hydrologic metric was evaluated for the 5-, 10-, and 20-year time periods to determine if it was a better predictor of macroinvertebrate response than was the $T_{0.5}$. Table 5-3 shows that the T_1 was a slightly better predictor of macroinvertebrate response than the $T_{0.5}$ for the 20-year time period (see Table 5-1), but was a worse predictor of

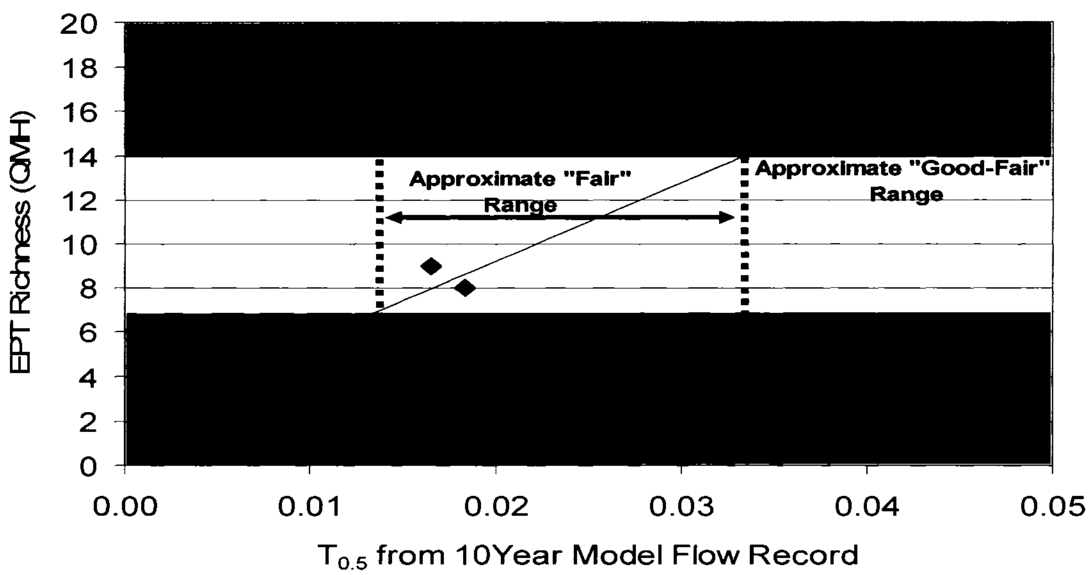


Figure 5-9. Relating $T_{0.5}$ Values for the 10-Year Flow Record to EPT Richness Rating Scores.

Table 5-3. Summary of Regression Results for the T_1 Hydrologic Metric.

Macroinvertebrate Metric	5-year	10-year	20-year	Average
EPT Rich (RTH)	0.55	0.71	0.79	0.68
EPT Rich (QMH)	**0.82**	**0.90**	**0.92**	0.88
EPT % Rich (RTH)	0.18	0.36	0.43	0.32
EPT % Rich (QMH)	0.68	0.79	0.82	0.76
B-IBI	0.57	0.65	0.69	0.64
Mean R^2	0.56	0.68	0.73	0.66

macroinvertebrate response than the $T_{0.5}$ for the 10-year time period. Overall, the T_1 hydrologic metric is a good predictor of EPT richness and percent richness.

The T_2 hydrologic metric was also evaluated to determine if it was a better predictor of macroinvertebrate response than the $T_{0.5}$ or the T_1. This metric was also evaluated to compare results computed from a partial duration series and those computed from an annual maximum series. Regression results shown in Table 5-4 demonstrate that the T_2 hydrologic metric was a better predictor of macroinvertebrate response than both the $T_{0.5}$ and the T_1 hydrologic metrics. Higher average correlations for both the partial

Table 5-4. Summary of Regression Results for the T_2 Hydrologic Metric.

	5-year		10-year		20-year		Average	
Macroinvertebrate Metric	Partial Duration	Annual Maximum	Partial Duration	Annual Maximum	Partial Duration	Annual Maximum	Partial Duration	Annual Maximum
EPT Rich (RTH)	0.91	0.88	0.83	0.94	0.69	**0.89**	0.81	0.90
EPT Rich (QMH)	**0.98**	0.94	**0.98**	**0.96**	**0.91**	0.83	**0.96**	**0.91**
EPT % Rich (RTH)	0.48	0.57	0.38	0.54	0.29	0.64	0.38	0.58
EPT % Rich (QMH)	0.92	**0.95**	0.86	0.87	0.77	0.87	0.85	0.90
B-IBI	0.73	0.63	0.75	0.83	0.67	0.65	0.72	0.70
Mean R^2	0.80	0.79	0.76	0.83	0.67	0.78	0.74	0.80

duration and annual maximum series were observed for all time periods examined for T_2 values. Interestingly, regression results for the T_2 metric from the five-year time period are much stronger than those for the $T_{0.5}$ hydrologic metric (see Table 5-1). Figure 5-10 shows regressions of the macroinvertebrate metrics against T_2 values computed from modeling results for the five-year time period for both the partial duration and annual

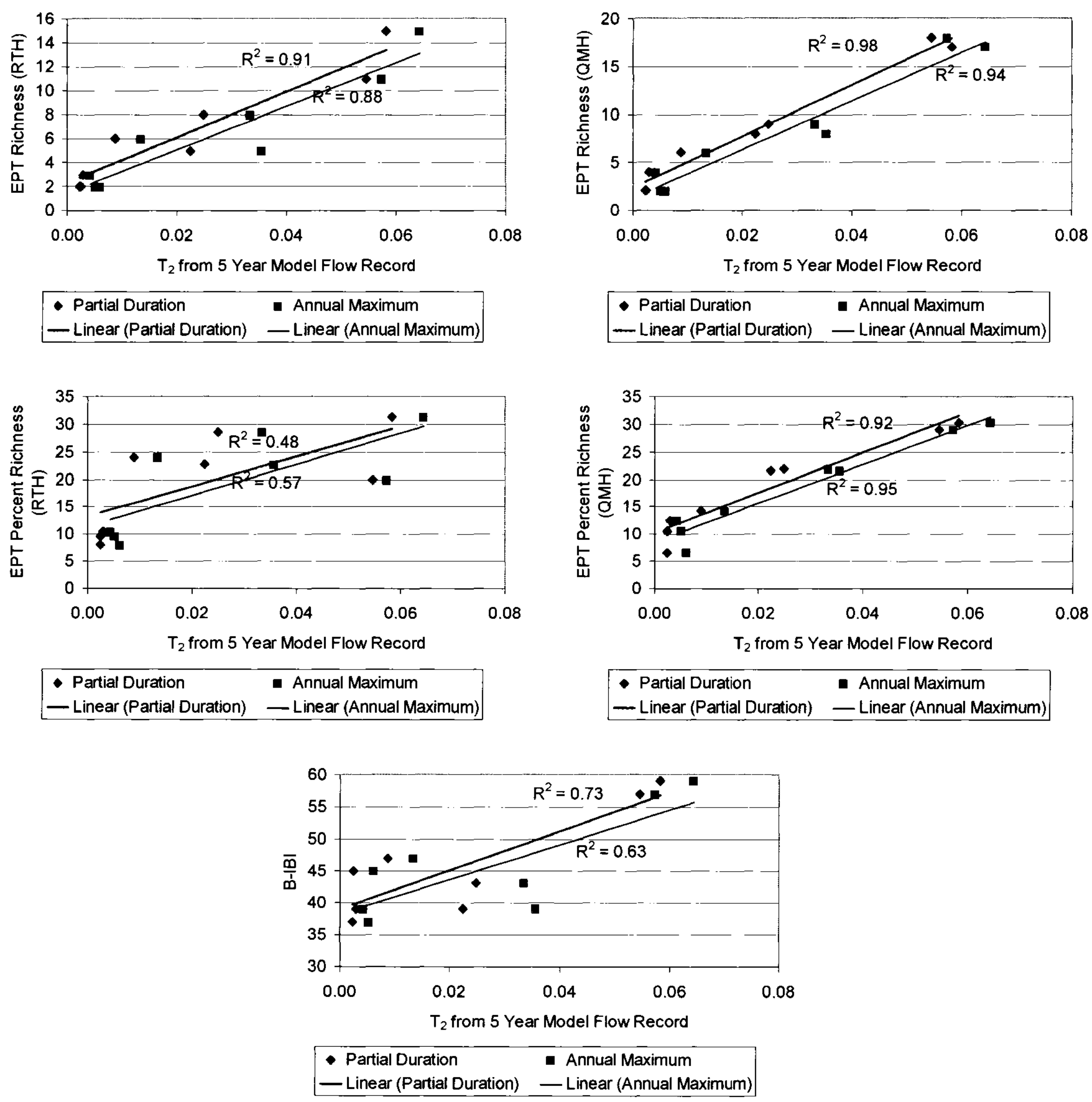


Figure 5-10. Relationship of T_2 Calculated From the 20-Year Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

maximum time series. Regression plots for the T_2 values computed for the 10- and 20-year time periods are shown Figures A-18 and A-19 in the Appendix.

Regression results for the partial duration T_2 hydrologic metric values for the 10- and 20-year time periods were similar to those for the $T_{0.5}$ hydrologic metric, however the T_2 hydrologic metrics computed from an annual maximum series for the 10- and 20-year time periods were better predictors of macroinvertebrate response. This indicates that an annual maximum series may be more appropriate for use for longer time periods. Similar average R^2 values were observed for the five-year time period.

Although Edgerly et al (2006) demonstrated that the T_{Qmean} hydrologic metric was not sensitive to stormwater controls, the metric was evaluated to determine its ability to predict biotic response, and its overall sensitivity to varied levels of urbanization in the eight North Carolina watersheds. This hydrologic metric was found to be valuable in predicting biotic response in the Puget Sound lowlands by both Booth et al. (2004) and Cassin et al. (2005). Unlike the $T_{0.5}$ hydrologic metric, however, Konrad et al. (2005) found that catchment size may influence T_{Qmean}. The influence of basin size on this hydrologic metric was not evaluated because the macroinvertebrate results for the eight watershed were not found to be related to the T_{Qmean}, as is shown in Table 5-5.

Table 5-5. Summary of Regression Results for the T_{QMean} Hydrologic Metric.

	1-year	2-year	5-year	10-year	20-year	Average
EPT Rich (RTH)	0.10	0.26	0.03	0.08	0.02	0.10
EPT Rich (QMH)	0.06	0.16	0.00	0.03	0.00	0.05
EPT % Rich (RTH)	0.08	0.15	0.01	0.02	0.00	0.05
EPT % Rich (QMH)	0.01	0.07	0.02	0.00	0.02	0.02
B-IBI	0.29	0.48	0.19	0.32	0.22	0.30
Mean R^2	0.11	0.22	0.05	0.09	0.05	0.10

5.2.2 Relationships Between Geomorphic Metrics and Biologic Parameters

Because it is known that biological assemblages and individual species are strongly associated with substrate size and condition (Waters, 1995; Berry et al., 2003), the effects of urbanization (UII) and the $T_{0.5}$ hydrologic metric on substrate size were evaluated for the study watersheds. Figure 5-11 shows the relationship between the predominant size class of bed substrate (see Table 4-3) evaluated by the USGS for each sampling location and the UII. The predominant size class was identified for each watershed as the size class that contained the highest percentage of samples with the dominant substrate in that size class. Figure 5-11 shows a logarithmic relationship between predominant size class and the UII.

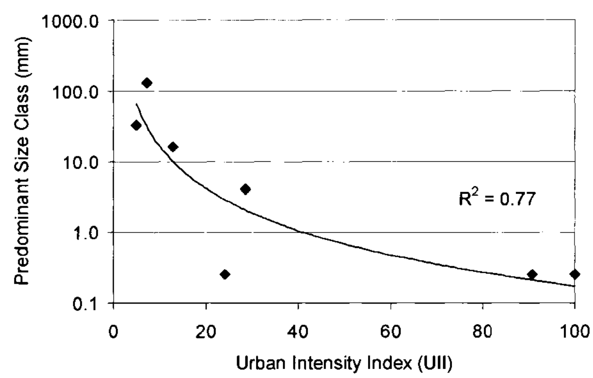


Figure 5-11. Relationship of Predominant Size Class of Substrate to Urban Intensity Index.

Figure 5-12 shows the relationship of the predominant size class of substrate to the $T_{0.5}$ hydrologic metric calculated from (a) the calibration period, (b) one-year model flow record, (c) two-year model flow record, (d) five-year model flow record, (e) 10-year model flow record, and (f) 20-year model flow record. Power function relationships were observed, with the best relationships occurring between the predominant size class and

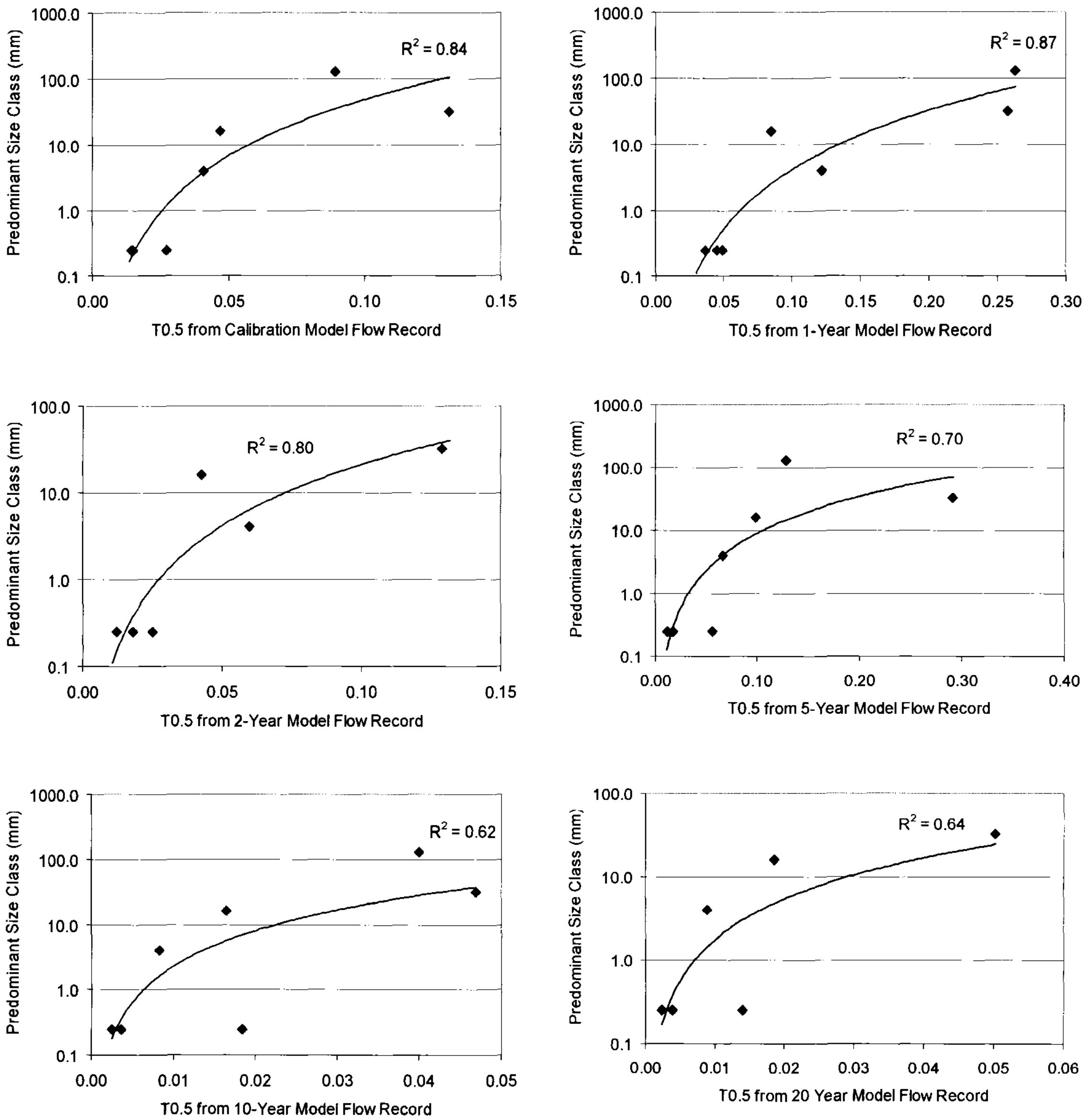


Figure 5-12. Relationship of Predominant Size Class of Substrate to $T_{0.5}$ Calculated From (a) the Calibration Period, (b) 1-Year Model Flow Record, (c) 2-Year Model Flow Record, (d) 5-Year Model Flow Record, (e) 10-Year Model Flow Record, and (f) 20-Year Model Flow Record.

the $T_{0.5}$ hydrologic metric calculated from the calibration and the one- and two-year time periods. This correlation indicates that the $T_{0.5}$ is viable as an indicator of predominant size class in these watersheds.

Figure 5-13 relates the predominant size class identified for each watershed to the corresponding macroinvertebrate score (see Table 4-3). Predominant categories ranged from sand to large cobble. As expected, the biologic scores are positively, logarithmically related to substrate size, with EPT Richness for the richest targeted habitat (RTH) having the strongest correlation with an R^2 of 0.90. It is not unexpected that the data collected for the RTH yielded the strongest correlation to substrate size because numerous studies have shown that biological assemblages and individual species are strongly associated with substrate size and conditions (i.e., Waters, 1995; Berry et al., 2003).

Figure 5-14 shows the relationship of the macroinvertebrate metrics to the cumulative excess shear stress computed for the one-year period of record prior to the benthic macroinvertebrate sampling. The cumulative excess shear stress over the one-year simulation period was calculated for the grain size of the predominant bed substrate at the sampling site of each watershed. Cumulative excess shear stress was not calculated for the Black Creek watershed because the predominant class consists of large boulders, irregular bedrock, irregular hardpan and irregular artificial surfaces does not have a definitive grain size. During the modeled 20-year time period no flows occurred in the Black Creek watershed that would move sediment with a grain size greater than 512-mm. It should also be noted that none of the flows that occurred in Strouds Creek were large enough to entrain the predominant substrate in that watershed, which was the size class large cobble, with a $d_s > 128$ mm.

Similar to the results relating predominant substrate to macroinvertebrate metrics, EPT richness showed the strongest relationships between cumulative excess shear stress from the 1-year time period examined and the benthic macroinvertebrate metrics. This is likely due to the influence of the substrate on habitat quality.

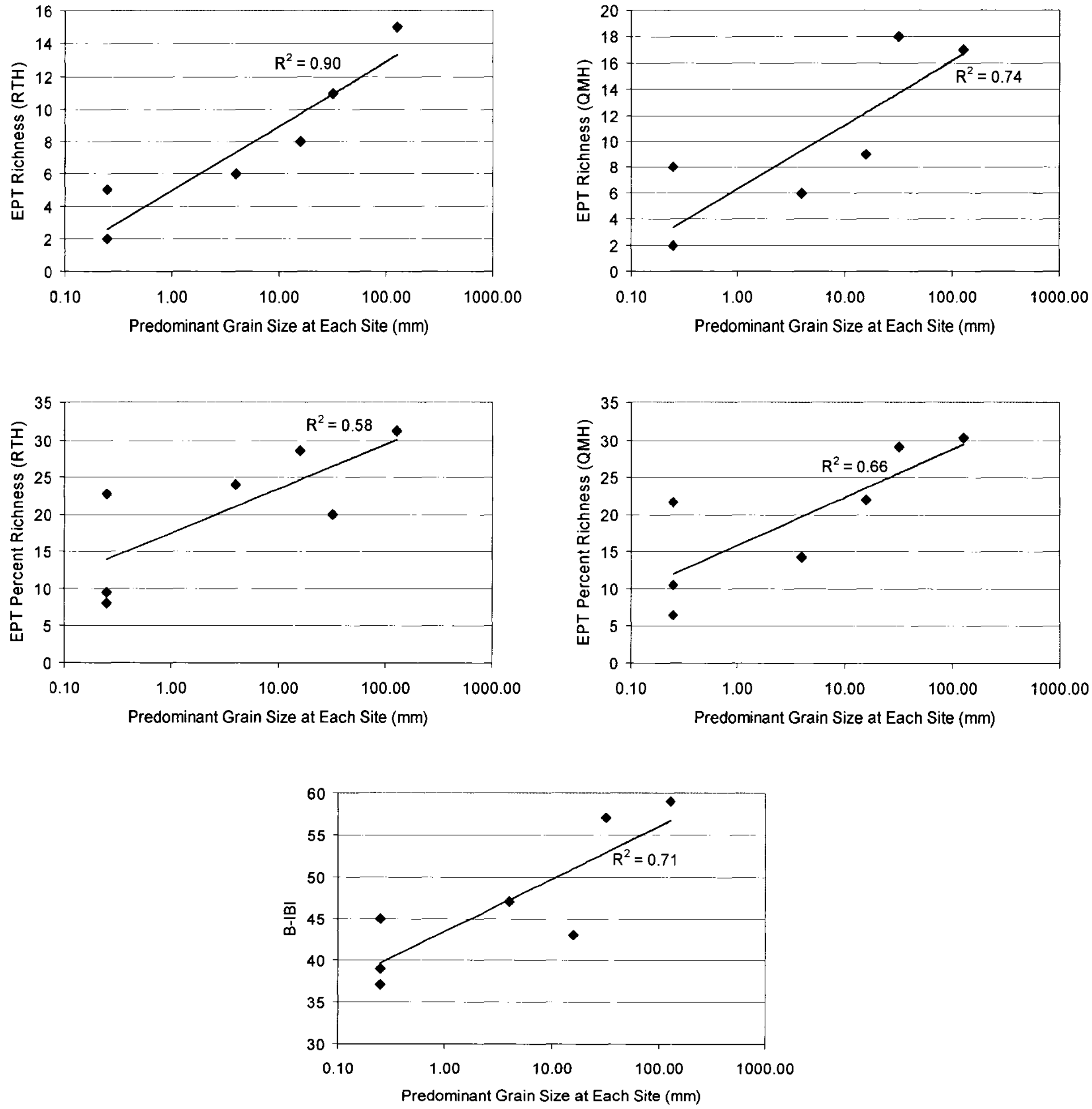


Figure 5-13. Relationship of Predominant Grain Size to (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

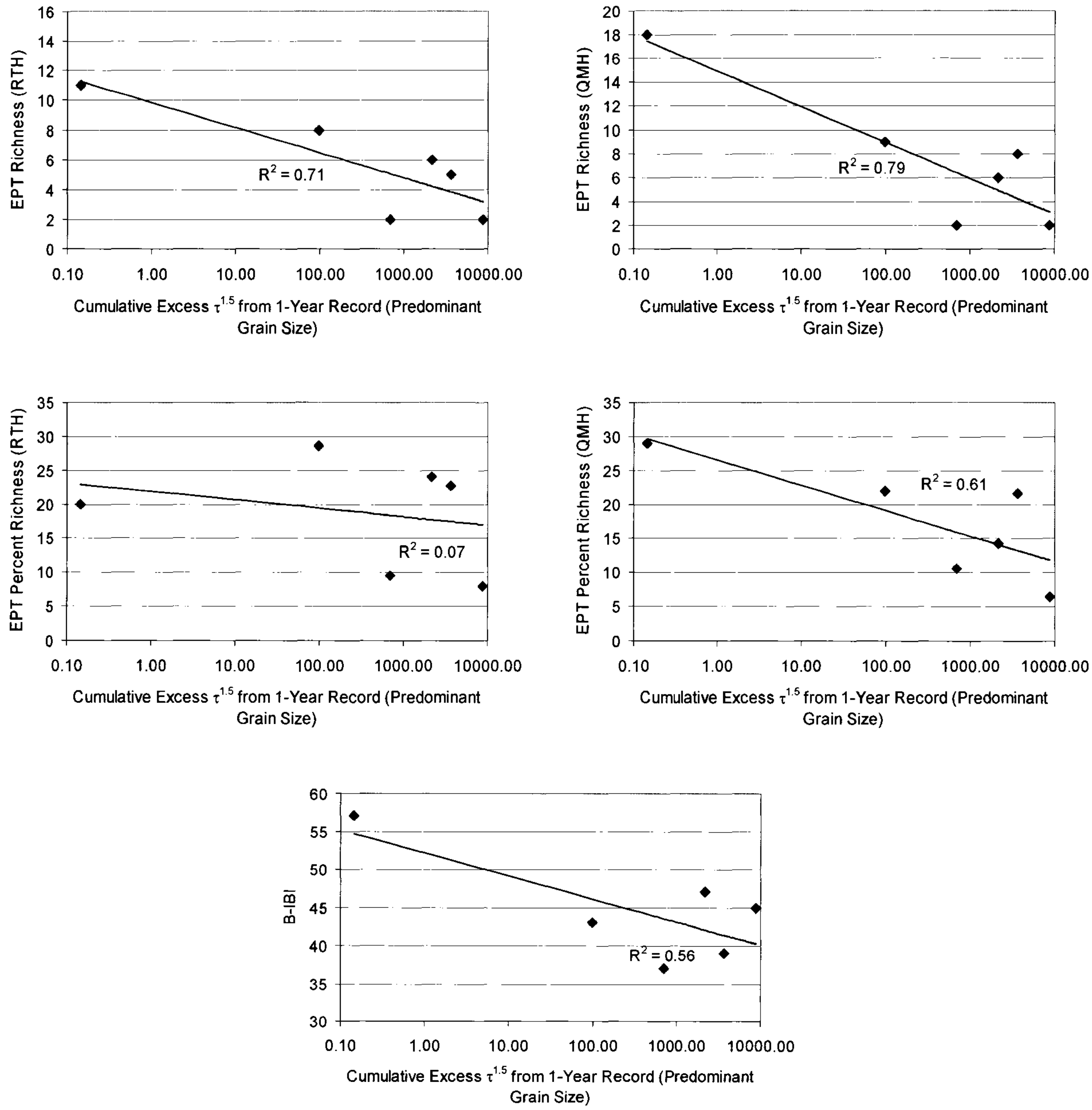


Figure 5-14. Relationship of Cumulative Excess Shear from the One-Year Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

When compared to the macroinvertebrate metrics, cumulative excess shear stress values computed for the two-, five-, 10- and 20-year time periods yielded similar regression results to those for the one-year time period. A summary of the R^2 values for each time period is provided in Table 5-6, which also summarizes the regression results for the macroinvertebrate metrics and the predominant substrate for each watershed shown in Figure 5-13. Overall, the predominant substrate for each watershed was a better predictor of macroinvertebrate response than was cumulative excess shear stress.

Table 5-6. Summary of Regression Results Between Macroinvertebrate Metrics and Geomorphic Metrics.

Macroinvertebrate Metric	R^2 Value						
	Cumulative Excess $\tau^{1.5}$						Predominant Substrate
	1-year	2-year	5-year	10-year	20-year	Average	
EPT Richness (RTH)	0.69	0.69	0.69	0.69	0.69	0.69	0.90
EPT Richness (QMH)	0.78	0.77	0.76	0.75	0.75	0.76	0.74
EPT % Richness (RTH)	0.06	0.06	0.08	0.07	0.07	0.07	0.58
EPT % Richness (QMH)	0.59	0.58	0.60	0.59	0.58	0.59	0.66
B-IBI	0.56	0.55	0.50	0.50	0.53	0.53	0.71
Mean R^2	0.54	0.53	0.53	0.52	0.52	0.53	0.72

Because the critical shear stress value assigned to each individual stream was determined by the grain size of the bed substrate, cumulative excess shear stress is not independent of the size of the bed substrate. It follows that the response of the macroinvertebrates to cumulative excess shear cannot be considered independently of the bed substrate. Because substrate was independently a better predictor of macroinvertebrate response than cumulative excess shear, this research shows that predominant substrate should be used as a geomorphic metric to predict macroinvertebrate response, rather than cumulative excess shear. The computation of cumulative excess shear, however, is necessary for determining erosion potential, which is shown to be a useful tool in evaluating stormwater management scenarios in Section

3.8.2. It should also be noted, however, that the hydrologic metrics ($T_{0.5}$, T_1, T_2) have as good of correlations between macroinvertebrate health as does the predominant substrate.

The relationship between cumulative excess shear stress and the predominant size class used to compute the cumulative excess shear stress is shown in Figure 5-15. Relationships between cumulative excess shear stress and the predominant size class were nearly the same for each time period examined, and were negatively related on a log-log scale. Independence of these two variables was not expected since the critical shear stress values used to calculate cumulative excess shear stress were determined based on the grain size of the predominant substrate size class.

5.2.3 Interrelationships between the $T_{0.5}$ hydrologic metric, substrate size, and macroinvertebrate response

In sections 5.2.1 and 5.2.2 it was shown that EPT richness and B-IBI values were related to both the $T_{0.5}$ hydrologic metric and substrate size as measured by the predominant size class found in each watershed. The observation of relationships between macroinvertebrate data and both hydrologic and geomorphic metrics leads to the question "Which variable is influencing the biotic response?" While based on the results of this study it is not possible to determine which variable more strongly influences biotic response, these results indicate that response is related to both variables. Additionally complicating the issue is the fact that Figure 5-12 showed that the predominant size class found in each watershed is correlated to the $T_{0.5}$ hydrologic metric, indicating that these two variables are not independent. Because these two variables are not independent, and because of the limited sample size (8 data points), multivariate regression was not

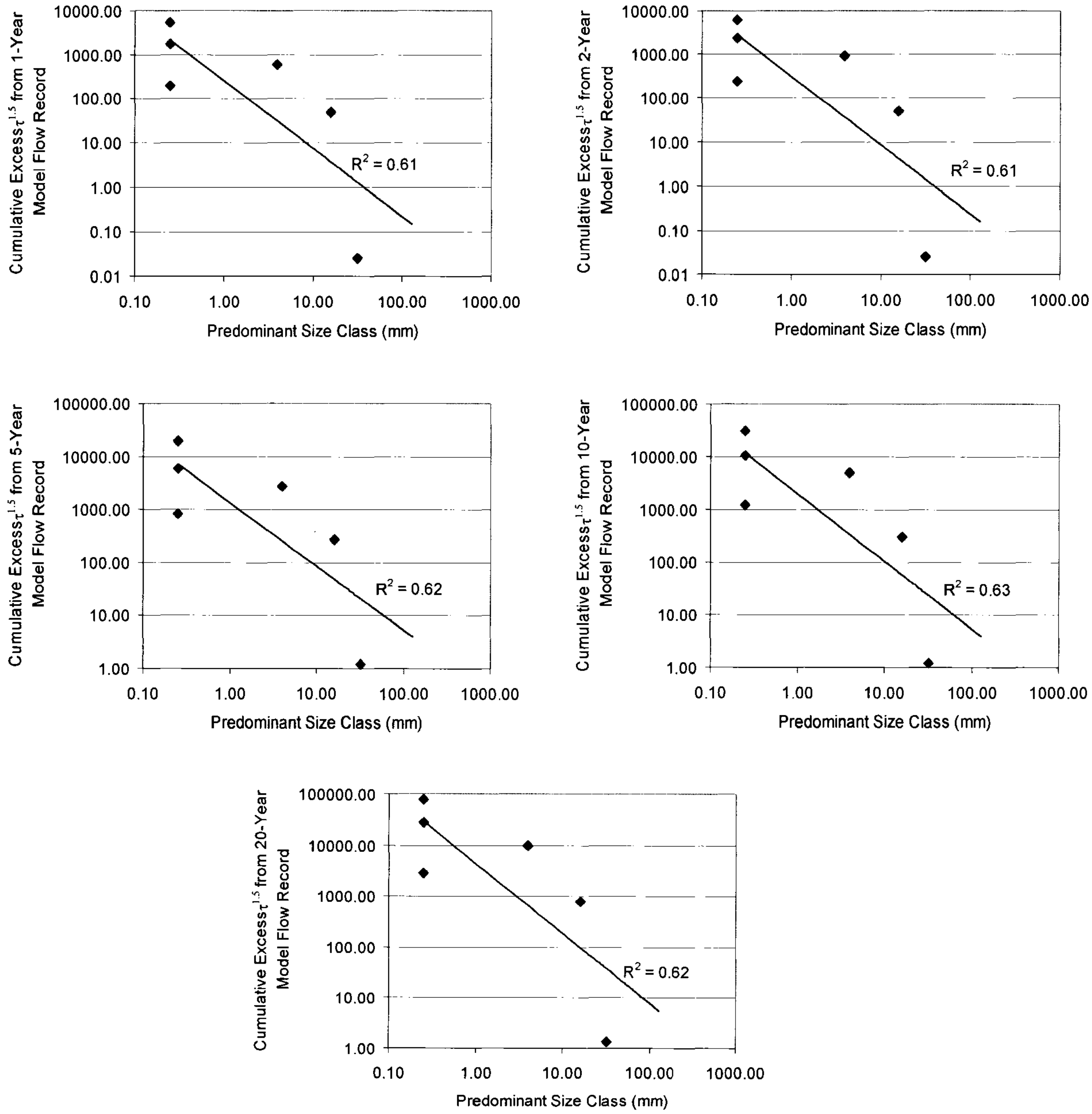


Figure 5-15. Relationship of the Predominant Size Class to Cumulative Excess Shear from the From (a) 1-Year Model Flow Record, (b) 2-Year Model Flow Record, (c) 5-Year Model Flow Record, (d) 10-Year Model Flow Record, and (e) 20-Year Model Flow Record.

conducted to determine the combined influence of these two factors in predicting biotic response.

Results from this study indicate that minimization of changes to the $T_{0.5}$, T_1 and T_2 hydrologic metrics will reduce changes in bed substrate critical for macroinvertebrate habitat. This is consistent with research by Osmundson (2002) that showed that flow regime can affect multiple aspects of an aquatic ecosystem, including habitat availability such as substrate. Controlling the $T_{0.5}$, T_1 and T_2 hydrologic metrics is likely to be beneficial in developing watersheds because it may reduce changes in bed substrate, but is likely to be less beneficial in restoration efforts of watersheds that are developed and have already experienced changes in bed substrate due to hydrologic alteration. The $T_{0.5}$ hydrologic metric may be a mechanistic way to describe relationships (linkages) between urbanization and biotic response, but may not be a stand alone linkage since bed substrate also appears to be a critical linkage between urbanization and biotic response.

5.3 Evaluating the Impact of Land Use Change and Runoff Control Strategies on Hydrologic and Geomorphic Metrics

The effects of land use and channel geometry on hydrologic and geomorphic metrics were evaluated in the Morgan Creek watershed using Step 6 of the protocol described in Section 3.8. Four scenarios were evaluated:

1. Rural land use in the watershed with rural channel geometry (existing conditions). Rural land use was modeled using existing land use in the watershed, calculated as described in Section 4.2.4,and varied by subcatchment from 0 - 11.8 % DCIA, with an overall average of 0.8 % DCIA

in the watershed. Channel geometry for rural conditions in the Morgan Creek watershed was based upon field measurements described in Section 4.2.1.

2. Urbanized land use in the watershed with rural channel geometry. Urbanized land use was based upon existing land use in the Pigeon House Creek watershed, which has an overall average of 14.3 % directly connected impervious area (DICA). A uniform DCIA value of 14.3 % was applied to all subcatchments in the Morgan Creek watershed for the urbanized land use scenarios. Channel cross-sections for this scenario were taken from Morgan Creek field measurements described in Section 4.2.1.

3. Urbanized land use in the watershed with urbanized channel geometry. A uniform DCIA value of 14.3 %, based on Pigeon House Creek watershed averages, was also applied to all of the Morgan Creek subcatchments in this scenario. Channel geometry for urbanized conditions was based on typical geometry corresponding to drainage areas of similar size in the Pigeon House Creek watershed. For example, the urbanized channel geometry applied in the Morgan Creek watershed was derived from the channel geometry representing a similar sized drainage area in the Pigeon House Creek watershed. Examples of cross-sections used for urbanized and rural channel geometry for the Morgan Creek watershed are shown in Figure 5-16.

4. Rural land use in the watershed with urbanized channel geometry. This scenario was evaluated for comparison purposes only; to give insight into whether streams in currently urbanized watershed could be returned to a predevelopment condition. Land use in this scenario was modeled using

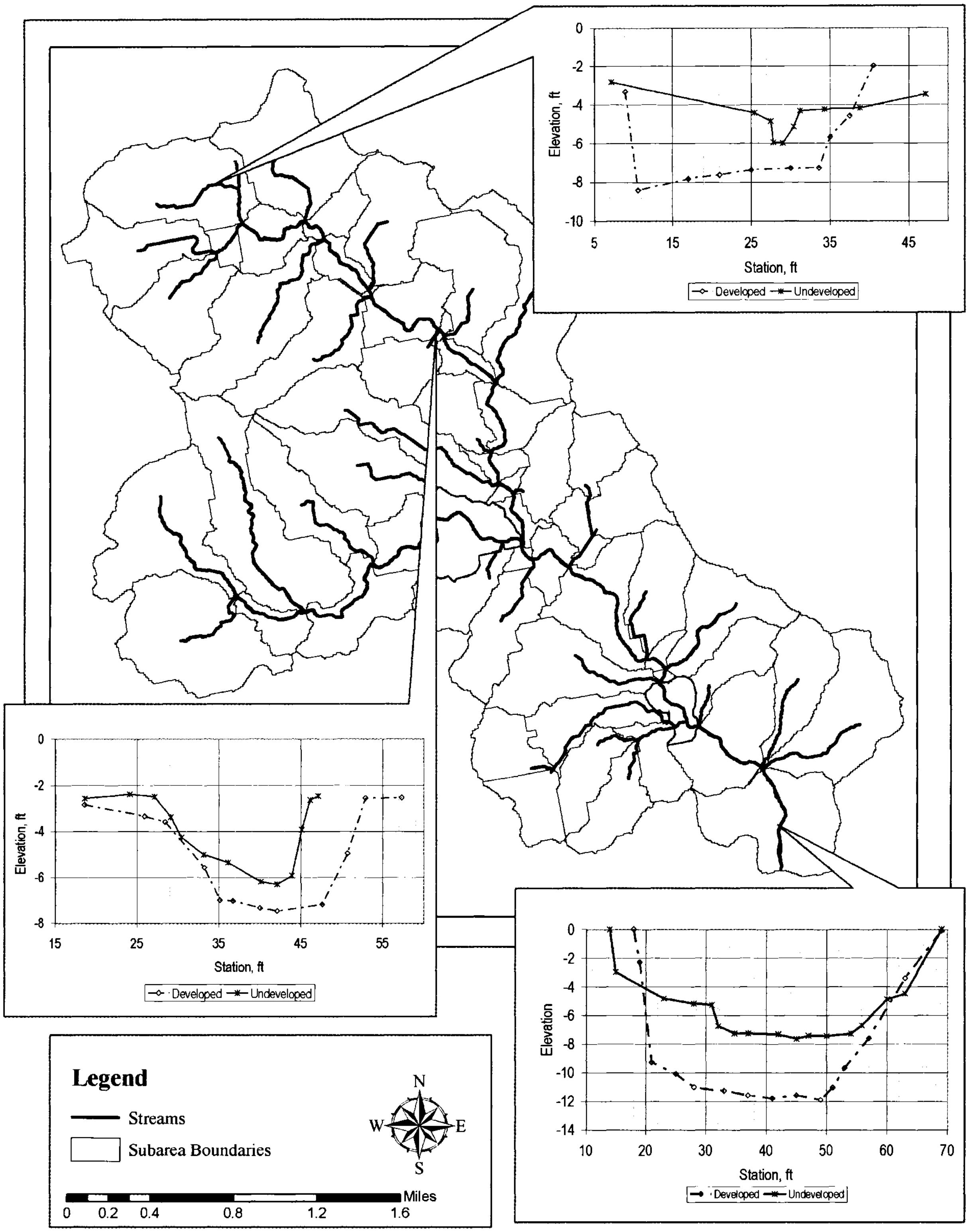


Figure 5-16. Representative Cross-Sections for Rural and Urbanized Channels in the Morgan Creek Watershed.

existing land use in the watershed, with an overall average of 0.8 % DCIA. Channel geometry for this scenario was based on typical geometry corresponding to drainage areas of similar size in the Pigeon House Creek watershed.

Application of urbanized channel geometry to the Morgan Creek watershed was not intended to predict channel evolution due to watershed land use changes; rather it was used to evaluate the relative impact of channel geometry on hydrologic metric values. Numerous variables influence channel erosion as the balance changes between water and sediment discharge, particle size and channel bedslope. However, enlargement of channels in the Morgan Creek watershed models is consistent with both anecdotal evidence and field research that support the notion that the larger and more frequent discharges that accompany watershed development cause downstream channels to enlarge (Wolman, 1967; Hammer, 1972; Leopold, 1973; Graf, 1975; Urbonas, 1980; Whipple et al., 1981; Neller, 1988, 1989; Booth, 1990; Caraco, 2000; Booth and Henshaw, 2001). Studies by Booth and Henshaw (2001) showed that rates of channel change in light to moderately urbanized watersheds in humid regions did not correlate with development intensity, but rather that the nature of the geologic substrate strongly influences whether or not significant channel change occurs. This finding is duly noted, however, the Morgan Creek and Pigeon House Creek watersheds are comprised of soils with similar simplified USDA soil texture classification (predominantly medium coarse to moderately fine) and hydrologic soil groupings (predominantly B & D, loam and clay), therefore the Pigeon House Creek watershed cross-sections were used as the best estimate available.

In addition to changes in land use and channel geometry, scenarios were investigated to examine changes in hydrologic and geomorphic metrics when stormwater controls are applied to the Morgan Creek watershed. Peak shaving detention and extended detention of the water quality capture volume (WQCV) was applied to 27 control points, as described in Section 4.2.4.

5.3.1 Hydrologic Analysis

The effects of land use and channel geometry on hydrologic and geomorphic metrics were evaluated in the Morgan Creek watershed. This section illustrates the hydrologic effects of land use changes, channel geometry changes, and the application of stormwater best management practices using peak flow frequency exceedence curves, flow duration curves, and the $T_{0.5}$ hydrologic metric.

Figure 5-17 and Figure 5-18 display the peak flow frequency exceedence curves for all of the scenarios evaluated in the Morgan Creek watershed. Existing conditions in the watershed are illustrated by the line depicting rural land use and rural channel geometry in Figure 5-17. As anticipated, the lowest peak discharge values occur under the rural land use, rural channel condition. The change in land use in the watershed from urban to rural creates an increase in peak discharge values across the full range of return periods when both rural and urban channels are simulated; a significantly larger increase in peak discharge values is observed when urban land use is combined with the urban channels. These increases in peak discharge are due to increased conveyance capacity throughout the system. When channels are widened and deepened, upstream attenuation of peak flows is reduced with decreases of overbank flooding and the loss of storage provided through floodplain connectivity. Figure 5-17 and Figure 5-18 also show the

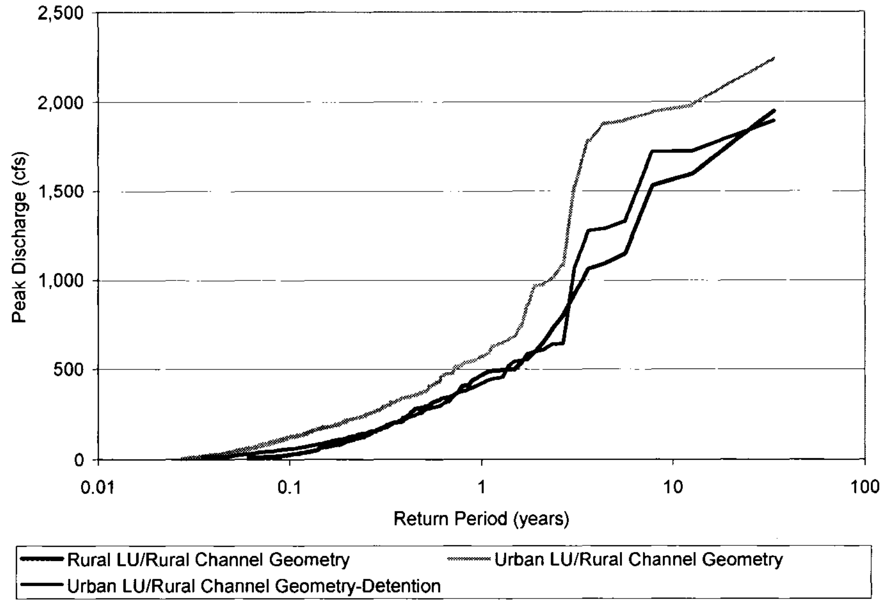


Figure 5-17. Effects of Detention on Peak Flow Frequency Exceedence Curves in Morgan Watershed with Rural Channels.

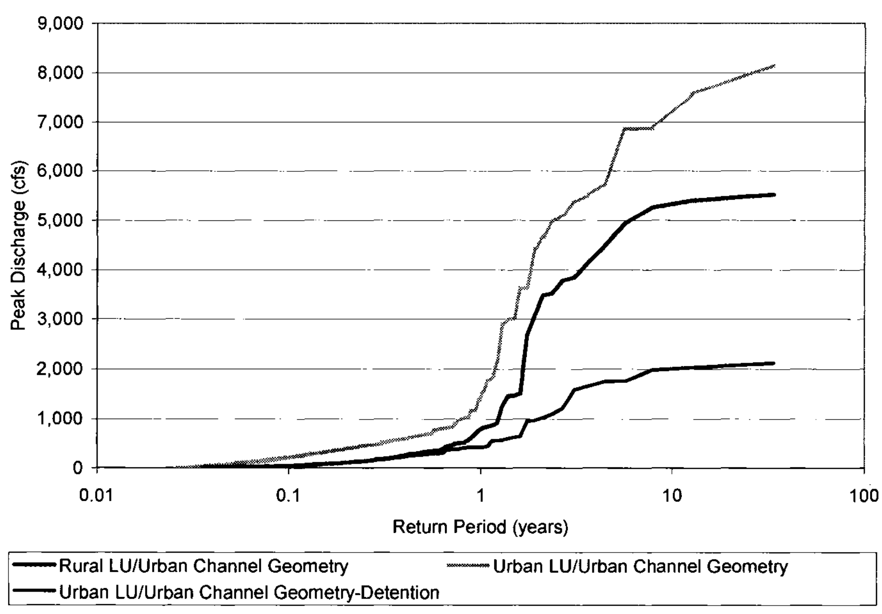


Figure 5-18. Effects of Detention on Peak Flow Frequency Exceedence Curves in Morgan Watershed with Urban Channels.

cumulative downstream effects of the 27 stormwater BMPs applied to the Morgan Creek watershed. Figure 5-18 demonstrates that the stormwater BMPs were designed to control peak discharges for rural channel geometry because the peak discharges are far below those for rural land use and urban channel geometry conditions.

Figure 5-19 and Figure 5-20 show the effects of land use change, channel geometry change, and stormwater management practices on the full spectrum of flows, as demonstrated by flow duration curves. Figure 5-19 shows that when runoff generated under urban conditions is controlled by stormwater BMPs, the magnitude and duration of large discharges (those that occur less than 1 percent of the time) closely matches those of rural conditions across a full spectrum of discharges. Changes in the durations of low flows due to the extended release of stormwater from the BMPs are not easily observed at the outlet of the watershed due to contributions of subsurface flows to baseflow in the model.

The $T_{0.5}$ hydrologic metric was computed for the four combinations of land use and channel geometry conditions modeled in the Morgan Creek watershed, as shown in Figure 5-21. Results are computed for the calibration period, one-year model flow record, two-year model flow record, five-year model flow record, 10-year model flow record, and 20-year model flow record. The effects of stormwater management practices on the $T_{0.5}$ are also shown.

Ranges of $T_{0.5}$ metric values for each of the three categories of EPT richness scores (poor, fair, and good-fair) were identified from the regression plots for each time period examined. Figure 5-9 illustrated this process. The colored panels in Figure 5-21 indicate the target ranges. It is important to note that the colored panels represent target

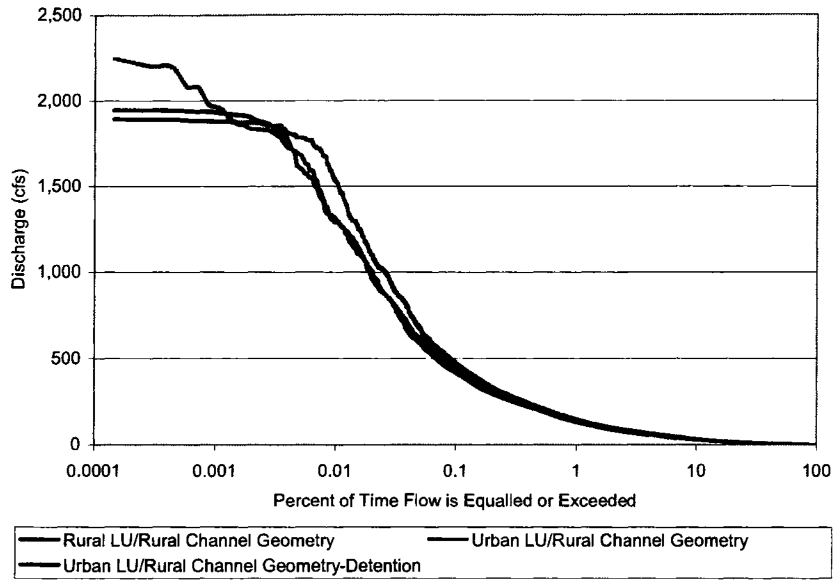


Figure 5-19. Effects of Detention on Flow Duration Curves in Morgan Watershed with Rural Channels.

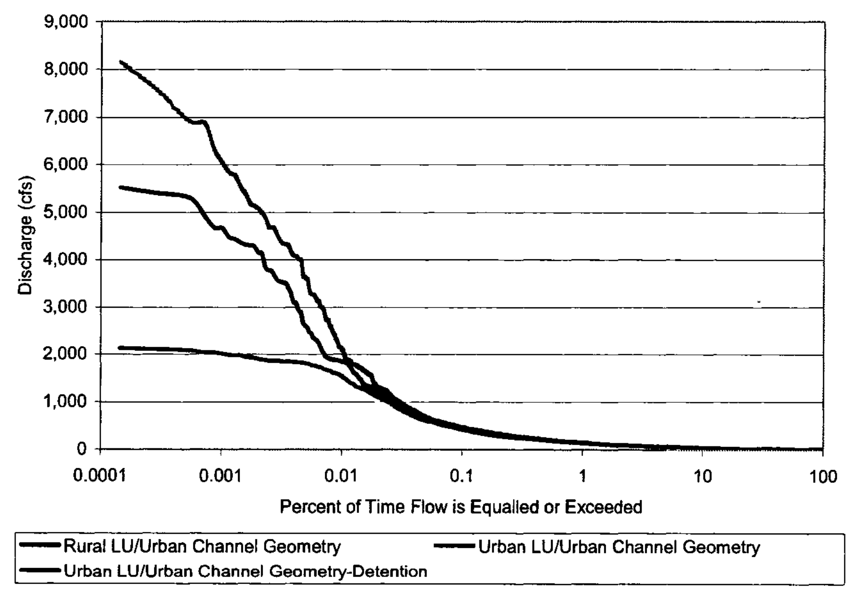


Figure 5-20. Effects of Detention on Flow Duration Curves in Morgan Watershed with Urban Channels.

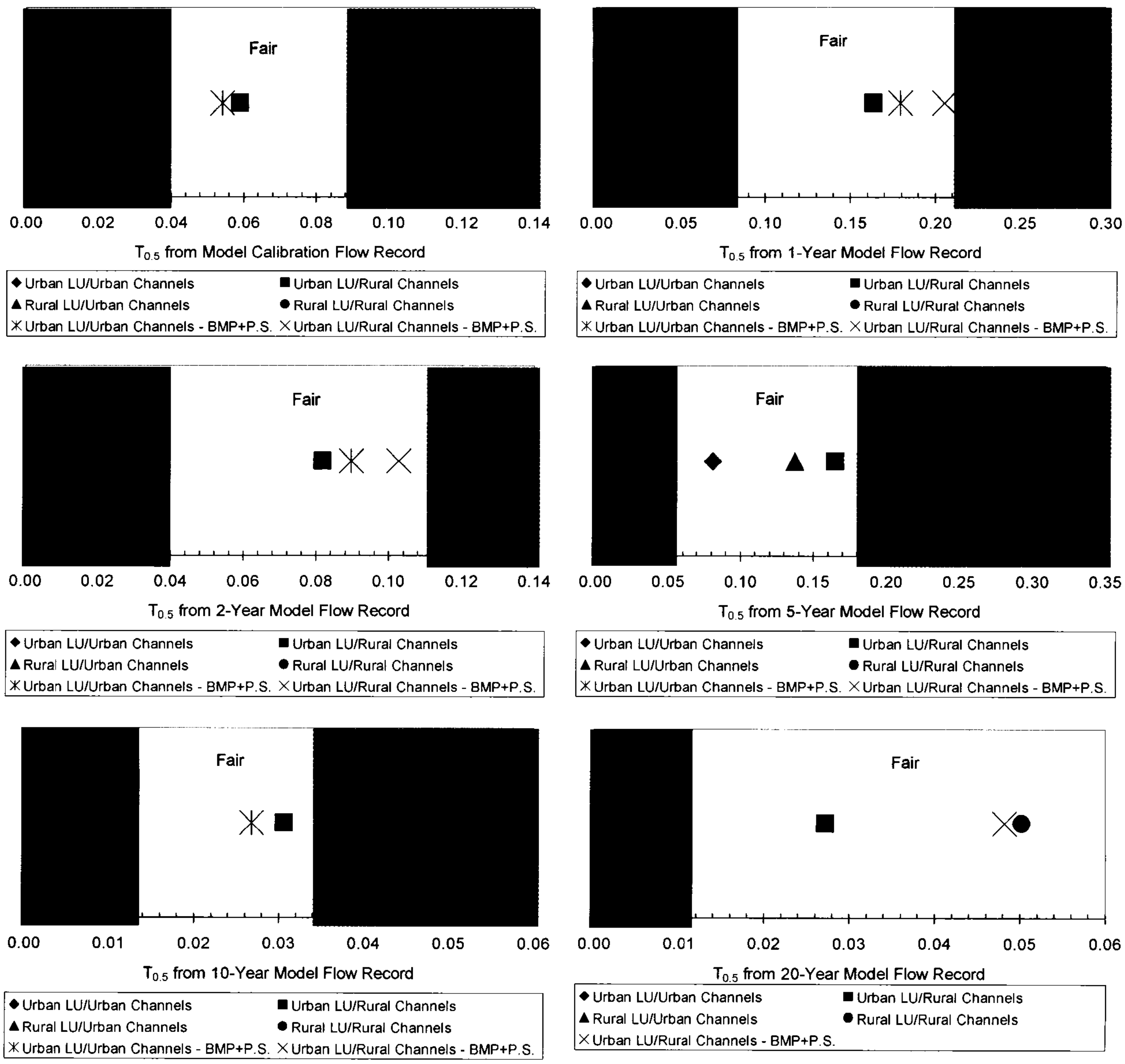


Figure 5-21. Effects of Land Use Change, Channel Geometry Change, and Stormwater Management Practices on the $T_{0.5}$ Metric in the Morgan Creek Watershed From the (a) the Calibration Period, (b) 1-Year Model Flow Record, (c) 2-Year Model Flow Record, d) 5-Year Model Flow Record, (e) 10-Year Model Flow Record, and (f) 20-Year Model Flow Record.

ranges, and are not meant to specifically identify thresholds, as there is uncertainty associated in the modeling efforts that were used to compute the $T_{0.5}$ metric, as well as uncertainty associated with the flow data used to calibrate the SWMM models, which was determined from the HEC-RAS models, measurements of stage data, and macroinvertebrate data which were collected at one point in time. Additionally, a number of factors other than the hydrologic flow regime play into macroinvertebrate health, such as water quality; these factors are not included in this analysis. Meeting the target ranges of $T_{0.5}$ metric values may not allow the target macroinvertebrate scores to be reached if other factors are at play.

Figure 5-21 shows that channel geometry plays a significant role in the flashiness of stream flows, as demonstrated by the $T_{0.5}$ metric values. In both urban and rural land use conditions, the loss of upstream flow attenuation creates flow conditions that could cause degradation in EPT richness. Consistent with the findings of Edgerly (2006), Figure 5-21 shows that application of stormwater controls in the Morgan Creek watershed affects the $T_{0.5}$ metric. The stormwater controls prevent the $T_{0.5}$ metric from decreasing to a value that would be in the range indicative of a poor EPT richness rating. The highest $T_{0.5}$ values were obtained when stormwater controls were applied in the urbanized watershed modeled with rural channels. This demonstrates the importance of preventing stream channel erosion and maintaining existing channel geometries when urbanization of a watershed takes place.

5.3.2 Geomorphic Analysis

This section demonstrates the influence of land use and stormwater management practices on instream erosion potential in the study watersheds. A full description of the methods used to compute erosion potential was presented in Section 3.6.2.4.

Table 5-7 shows the effect of stormwater management practices on erosion potential using cumulative excess shear stress values to the 1.5 power for very coarse gravel (d_s>32 mm). The predominant size classes in the Morgan Creek watershed are very coarse gravel and small cobble (d_s>64 mm), each class individually accounted for 22% of the samples evaluated. Critical shear stress for the small cobble was used in this analysis. The erosion potential analysis was conducted for rural channels, only, as it is not appropriate to relate cumulative excess shear stress values computed for channels with different geometry. Table 5-7 shows that when stormwater controls are applied to the Morgan Creek watershed a desired erosion potential value of 1.0 is achieved when computed for the 20-year period of record; a slight over control of erosion potential occurs for the 5- and 10-year periods of record. This over control was expected, given that the flow duration curves shown in Figure 5-19 showed that the high-magnitude flows at the outlet of the Morgan Creek watershed were slightly over controlled due to the downstream combined effects of the 27 stormwater basins.

The high erosion potential values computed for the one- and two-year time periods occur because the cumulative excess shear values for those time periods were extremely small because very few flows occurred that were large enough to entrain the substrate. It is more appropriate to evaluate erosion potential for a longer period of time, when a larger spectrum of flows is more likely to occur. Relatively low erosion potential values are observed for the 5-, 10-, and 20-year time periods when the Morgan Creek watershed was developed without stormwater controls. This is because the predominant

Table 5-7. Summary of Erosion Potential for the Morgan Creek Watershed.

Scenario	1-Year	2-Year	5-Year	10-Year	20-Year
Developed, No Controls	12.54	12.54	1.09	1.24	1.54
Developed, BMP+Peak Shaving	4.37	4.37	0.93	0.93	0.99

substrate in the stream channel is small cobble, and the flows required to entrain this substrate are large. The peak flow frequency exceedance curve shown in Figure 5-17 demonstrated that increases in the magnitude and frequency of the highest peak discharge values for the 20-year simulations were not substantial, therefore it is expected that erosion potential for small cobble would not be substantial. Erosion potential for other substrates, however, would be affected more significantly, as was demonstrated in the Fort Collins, Colorado and Lenexa, Kansas examples summarized in Sections 3.8.2.1 and 3.8.2.2 .

It should be noted that the erosion potential computations do not account for changes in upland sediment supply that occur with land use change in a watershed or the application of stormwater management BMPs. Stream channel stability is dependant on a balance between sediment transport capacity of a stream and the supply of sediment both from upland and in-stream sources, therefore erosion potential values of less than 1.0 do not necessarily mean that the flows in the channel are overcontrolled.

6.0 SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

6.1 Summary

A protocol was presented that stormwater managers can use to evaluate how alternative land use development patterns and runoff controls will impact the biotic health of receiving streams in urbanizing watersheds. The protocol describes the hydrologic and biologic data required evaluate these impacts, methods to synthetically generate the hydrologic data if it is not available, and the analytical procedures necessary for data analysis. A study of eight watersheds in the North Carolina Piedmont with a gradient of urban development ranging from fully urbanized to rural was conducted and demonstrated that the $T_{0.5}$ and other similar hydrologic metrics are responsive to changes in land use. The $T_{0.5}$ hydrologic metric was also found to be responsive to alternative runoff control scenarios. It was also found that the biotic health of streams in the North Carolina Piedmont, as measured by macroinvertebrate indices, is responsive to the $T_{0.5}$ metric, a finding similar to that found by Booth et al. (2004) in a study of the streams in the Pacific Northwest, a region of completely different hydrology. Macroinvertebrate metrics were also responsive to the T_1 and T_2 metrics.

It was demonstrated that limited flow monitoring and macroinvertebrate data could be used in conjunction with hydrologic and hydraulic modeling (through SWMM5) to estimate how the biotic integrity of streams in a developing watershed might be affected through changes in land use and runoff control scenarios. The $T_{0.5}$ hydrologic metric values computed from the model were linearly related to those computed from the flow monitoring data provided by the USGS. While the values were not identical, the linear relationship indicates that use of the SWMM5 model for computation of this metric is useful and appropriate. The $T_{0.5}$ hydrologic metric values computed from the model data were logarithmically related to the urban intensity index for all time periods examined (calibration period, 1-, 2-, 5-, 10-, and 20-year). The $T_{0.5}$ hydrologic metric was not found to be related to watershed drainage area.

Macroinvertebrate metrics (EPT richness, EPT percent richness, and B-IBI) were linearly related to the $T_{0.5}$ hydrologic metric computed from both the gage flow record and the flow record generated by the model. Poor relationships between EPT percent richness measured for the richest targeted habitat and the $T_{0.5}$ hydrologic metric were observed for all time periods examined. Of the macroinvertebrate metrics, overall, EPT richness for the qualitative multihabitat was most strongly related to the $T_{0.5}$ hydrologic metric. For the 1- and 2-year time periods, the B-IBI metric had a slightly stronger relationship to the $T_{0.5}$ hydrologic metric. The $T_{0.5}$ hydrologic metric values computed from the 1- and 2-, and 10-year time periods were the best predictors of macroinvertebrate metric scores; the 5-year time period was the worst, likely due to the influence of Hurricane Floyd on the $T_{0.5}$ hydrologic metric values computed for this time period. Macroinvertebrate metric values were better predicted by the urban intensity index (UII) than the directly connected impervious area of the watershed, but the UII was a weaker predictor of macroinvertebrate metric values than the $T_{0.5}$ computed from the model for several of the time periods examined. For the watersheds in this study, $T_{0.5}$ hydrologic metric values computed from both 15-minute and hourly time steps were good predictors of macroinvertebrate metric values, however relationships between the macroinvertebrate metrics and the $T_{0.5}$ hydrologic metric values were always stronger when computed from the 15-minute data. $T_{0.5}$ hydrologic metric values computed from daily time steps are not predictors of macroinvertebrate response.

The T_{Qmean} hydrologic metric was not a predictor of macroinvertebrate response in this study. The T_1 hydrologic metric did not predict macroinvertebrate response as well as the $T_{0.5}$ for the 5- and10-year time periods, but did predict macroinvertebrate response slightly better then the $T_{0.5}$ for the 20-year time period. The T_2 hydrologic metric, however, was a better predictor of macroinvertebrate response than both the $T_{0.5}$ and the T_1 hydrologic metrics. T_2 hydrologic metrics computed from an annual maximum series, rather than a partial duration series, were better predictors of macroinvertebrate response for the longer (10- and 20-year) time periods, but offered approximately the same level of prediction when computed for the 5-year time period.

The $T_{0.5}$ hydrologic metric was affected by both land use and channel geometry. When land use in the Morgan Creek watershed was changed from rural to urban, but the channel geometry remained the same, the value of the $T_{0.5}$ hydrologic metric did not decrease as much as when both land use and channel geometry were changed. When the channel geometry of the streams modeled for the Morgan Creek watershed were changed to be similar to those observed in urban watersheds of the North Carolina Piedmont, but land use remained the same (rural), the $T_{0.5}$ hydrologic metric was nearly the same as the one computed when both land use and channel geometry were changed. This indicates that channel geometry plays a significant role in values of the $T_{0.5}$ hydrologic metric and that channel evolution due to incision and stream bank erosion should be prevented when urbanization takes place in order to minimize negative biotic response. Prevention of stream channel incision and erosion can occur if the balance of sediment supply and transport capacity in the stream channel is maintained from pre- to post-development conditions. If this balance is not achievable, stream bank stabilization measures (i.e., armoring, drop structures) may be necessary to limit channel degradation.

Results of the $T_{0.5}$ hydrologic metrics computed for the eight North Carolina Piedmont watersheds were compared to qualitative multihabitat EPT richness values to identify approximate ranges of $T_{0.5}$ scores that yielded good-fair, fair, and poor EPT richness ratings. These ranges of $T_{0.5}$ scores were used to evaluate the potential biotic impact of changes in land use and channel geometry in the Morgan Creek watershed, as well as to evaluate the potential impacts of stormwater management practices. Modification of channel geometry to be similar to cross-sections observed in urban watersheds changed the $T_{0.5}$ hydrologic metric enough to cause it to predict a degraded EPT richness score of poor, except in the 5-year time period, which was not shown to a good of a predictor of macroinvertebrate response. When stormwater controls were applied to the developed Morgan Creek watershed, and when channel geometry was left in its original state, changes in the value of the $T_{0.5}$ hydrologic metric were minimized, and anticipated changes to macroinvertebrate ratings were reduced, albeit not eliminated. When stormwater controls were applied to a developed Morgan Creek watershed with urbanized channels the $T_{0.5}$ hydrologic metric value was greatly improved, but not sufficiently enough to minimize changes in the macroinvertebrate score, which was almost always reduced to fair.

The predominant size class of substrate measured for each watershed was logarithmically related to the urban intensity index. The predominant size class of substrate related to the $T_{0.5}$ hydrologic metric by a power function; strong relationships were observed for the most recent time periods (calibration, 1- and 2-year time periods) and weaker relationships were observed between the 5-, 10- and 20-year time periods. Macroinvertebrate metrics were logarithmically related to predominant substrate size, with EPT richness for the richest targeted habitat having the strongest correlation. Cumulative excess shear stress was a worse predictor of the macroinvertebrate metrics than was the predominant grain size, except for EPT richness for the qualitative multi-habitat results. Similar relationships were observed between the macroinvertebrate metrics and the cumulative excess shear stress computed for each time period examined. Long-term changes in erosion potential in the Morgan Creek watershed were minimized when stormwater controls were applied.

6.2 Conclusions

Hydrologic and geomorphic metrics computed from continuous, long-term, instream flow records were shown to be useful in establishing linkages between stormwater management practices and stream ecosystem health as measured by macroinvertebrates. Establishment of these linkages can allow municipalities to relate land use planning and runoff control strategies to ecologic health in streams, and thus helps them identify stormwater management criteria that will provide protection of urban streams.

Rainfall/runoff models can be used to generate the stream flow data needed for the computation of hydrologic and geomorphic metrics for the purposes of evaluating the effects of stormwater management practices. Model data can be used to supplement instream flow monitoring conducted by the USGS and other entities, which may not be available long-term for watersheds at a scale that allows for the evaluation of the effect of stormwater management practices on hydrologic and geomorphic metrics.

Use of rainfall/runoff models allows for generation of stream flow data needed for the computation of hydrologic and geomorphic metrics in linkage development; however, these models should be used with caution for the computation of hydrologic and geomorphic metrics of biologic interest. Many factors influence historical or natural flow regimes such as large woody debris, beaver dams, wetlands, or changes in floodplain connectivity. Additionally, other factors can influence current and future flow such as the influence of groundwater withdrawls on summer baseflows or point source discharges from wastewater treatment plants. These factors may not be accurately estimated by models. Ease in calibration of the SWMM5 models used in this study to high flows indicate that the SWMM5 model is appropriate for computation of hydrologic metrics such as the $T_{0.5}$, T_1, or T_2, because these metrics are based on durations of high flows. The model, however, is not recommended for computation of low flow metrics such as the low pulse duration or low pulse range described by Cassin et al. (2005), due to difficulties calibrating the model to baseflows and low flows.

Hydrologic metrics, which reflect the change in the hydrologic regime that occurs with urban development, have a higher correlation to the biologic health of a stream, as measured by macroinvertebrate indices, than indirect measures of urbanization such as percent imperviousness. These metrics are useful descriptors of urbanization because they can be used to identify targets for control of stormwater runoff.

It is difficult to discern whether biotic response is more dependent on hydrologic or geomorphic metric values, because the two are not independent. This classic "chicken and egg" question is important in terms of use of this protocol for stream restoration. Restoration of hydrology, alone, may not allow for the attainment of biotic goals if habitat (as described by geomorphic metrics such as substrate size or erosion potential) is not also restored. In urbanizing situations, however, preservation of a more natural hydrology through the use of hydrologic metrics and restrictions on increases in erosion potential will reduce geomorphic changes from what would occur if stormwater management controls were not implemented. In addition to negative influences of changes in substrate on biotic response, minimization of channel erosion is also important in terms of its influence on hydrologic metrics. Channel geometry directly influences hydrologic metric values due to its impact on conveyance capabilities and loss of flood attenuation due to disconnectivity from floodplains.

In addition to evaluating the hydrologic impacts of land use change and application of stormwater controls in a watershed, it is important to also account for changes in upland sediment supply when evaluating erosion potential. Stream channel stability is dependant on a balance between sediment transport capacity of a stream and the supply of sediment both from upland and in-stream sources.

The protocol presented herein is not the last word in evaluating the impacts of urban runoff on the biotic integrity of streams, but rather it is a tool or methodology that can be used to help stormwater managers and planners to better understand the potential impacts of implementing alternative structural runoff control strategies on stream in urbanizing watersheds. It does not predict or simulate the biotic health of streams, but rather provides an estimate for the relative degradation in the biotic integrity of streams that might be expected if certain runoff control strategies are, or are not implemented.

In small communities that cannot afford to conduct the data sampling and computer modeling required by the protocol, the following set of minimum runoff control criteria for new development can be applied to provide improved protection of streams from a hydrologic, geomorphic, and biologic perspective:

- The peak discharge from the 100-, 10- and 2-yr storm shall not exceed the peak discharges for these storms in the pre-development condition.
- An extended detention pond with a 40-hr drawdown time that is sized according to the WEF and ASCE (1998) Manual of Practice on Urban Stormwater Management shall be required on new development.

6.3 Recommendations

The protocol is a step forward in developing tools that municipalities can use to determine the effect of their runoff control practices on the biotic health of the streams in urbanizing watersheds. There is a large body of literature on the effects of urbanization on the geomorphology and the biotic integrity of receiving streams, but there are limited tools that stormwater managers can use to help them determine the best stormwater management practices to require of developers to minimize the effects of urbanization on the receiving streams. The protocol attempts to fill that knowledge gap. As additional knowledge is gained on the cause-effect relationships of flow, geomorphology, water quality and biotic integrity in urban streams, these relationships can be incorporated into the protocol, improving its predictive capability and usefulness as a planning tool. Also helpful for this protocol would be the development of publicly available tools that compute hydrologic metrics for varied timesteps (15-minute or hourly), through the expansion of NATHAT (Henrickson, 2006) , the IHA (The Nature Conservancy, 2007) or GeoTool (Bledsoe, et al., *in press*).

It should be borne in mind that the watersheds in the North Carolina Piedmont used in this study do not have properly sized BMPs or runoff controls for small storms; nor do the watersheds reviewed in the literature as far as is known. It is recommended that a controlled experiment be conducted on an urbanizing watershed of 1 - 3 square miles, with runoff controls and BMPs designed according to the criteria recommended in the study presented herein. This study can relate macroinvertebrate health to hydrologic metrics as the watershed develops and compare resulting data to watersheds that are in various states of urbanization, but that have no significant runoff controls. Care must be taken in this experiment to control runoff during the construction phase or the macroinvertebrate indices will reflect the construction impacts rather than the impacts of the built environment.

Further research is needed on the effects of urbanization and stormwater management practices on watershed sediment load, and the potential impacts of these changes on channel erosion potential and channel geometry.

Lastly, water quality was not addressed in this work. From the standpoint of protecting the biotic integrity of receiving streams in urbanizing watersheds it is important to minimize the interruption of the natural hydrologic cycle that causes geomorphic and biologic degradation in stream channels *and* to address water quality. Runoff controls required to achieve hydrologic and geomorphic stability must include water quality control BMPs, which if properly designed will remove pollutants from stormwater runoff. The federal Clean Water Act is based on water quality criteria in receiving streams and stormwater regulations now require that Total Maximum Daily Loads (TMDLs) include stormwater runoff. Therefore, it is recommended that future research build on this protocol, adding sampling of constituents identified in the water quality criteria applicable to the receiving stream, and that stormwater models add water quality as a runoff parameter and simulate the removal of pollutants by runoff control practices. Further development of currently available rainfall runoff models, including the SWMM5 model, is needed to be able to credibly simulate instream runoff quality and constituent removal in stormwater management practices.

7.0 REFERENCES

Akan, A.O. and Houghtalen, R.J. 2003. Urban Hydrology, Hydraulics, and Stormwater Quality: Engineering Applications and Computer Modeling. John Wiley & Sons, Inc.

Alberti, M., Booth, D.B., Hill, M., Marzluff, J. 2003. *The Impacts of Urban Patterns on Ecosystem Dynamics* National Science Foundation-Research Grant. Final Report.

Allan, C. and Curtis, A. 2005. Nipped in the Bud: Why Regional Scale Adaptive Management Is Not Blooming. *Environmental Management* 36(3):414–425.

Arcement, G.J. and Schneider, V.R. 1989. Guide for selecting Manning's roughness coefficients for natural channels and flood plains. U.S. Geological Survey Water Supply Paper 2339.

Barbour M.T. and Yoder, C.O. 2005. *Technical Guidelines: Critical Technical Elements For An Effective Bioassessment Program.* 2005 Draft in preparation.

Barbour, M.T., Paul, M.J., Bressler, D.W., Purcell, A.H., Resh, V.H., and Rankin, E. *In press. Water Environmental Research Foundation Project No. 01-WSM-3. Research Digest: Bioassessment: A Tool for Managing Aquatic Life Uses for Urban Streams* (DRAFT).

Berry, W., Rubinstein, N. Melzian, B., and Hill, B. 2003. T*he Biological Effects of Suspended and Bedded Sediment (SABS) in Aquatic Systems: A Review. U.S. EPA. Internal Report, August 20, 2003.* Office of Research and Development.

Bicknell, B.R., Imhoff, J.C., Kittle, J.L., Jr., Donigian, A.S., Jr. and Johanson, R.C. 1997. *Hydrologic Simulation Program – Fortran: User's Manual for Release 11*, USEPA, Office of Research and Development, Athens, GA, (Available from: http://www.epa.gov/ceampubl/softwdos.htm.)

Bledsoe, B.P. 2002. Stream Erosion Potential and Stormwater Management Strategies. *Journal of Water Resources Planning and Management*, 128:451-455.

Bledsoe, B.P. and Watson, C.C. 2001. Effects of urbanization on channel instability. *Journal of the American Water Resources Association* 37(2):225-270.

Bledsoe, B.P., Brown, M.C., Raff, D.A. In Press. GeoTools: A Toolkit for Fluvial System Analysis. Accepted by *Journal of the American Water Resources Association.*

Booker, D.J. and Dunbar, M.J. 2004. Application of Physical Habitat Simulation (Phabsim) Modeling to Modified Urban River Channels. *River Research and Applications* 20(2):167-183.

Booth, D.B. 1990. Stream-Channel Incision Following Drainage-Basin Urbanization. *Journal of the American Water Resources Association.* 26:407-417.

Booth, D.B., Hartley, D., and Jackson, R. 2002. Forest Cover, Impervious-Surface Area, and the Mitigation of Stormwater Impacts. *Journal of the American Water Resources Association* 38(3): 835-845.

Booth, D.B. and Jackson, C.R. 1997. Urbanization of Aquatic Systems: Degradation Thresholds, Stormwater Detection, and the Limits of Mitigation. *Journal of the American Water Resources Association* 33(5):1077-1090.

Booth, D.B., Karr, J.R., Schauman, S., Konrad, C.P., Morley, S.A., Larson, M.G., and Burger, S.J. 2004. Reviving Urban Streams: Land Use, Hydrology, Biology, and Human Behavior. *Journal of the American Water Resources Association* 40(5):1351-1364.

Booth, Derek B. and P.C. Henshaw. 2001. "Rates of Channel Erosion in Small Urban Streams." In *Land Use and Watersheds, Human Influence on Hydrology and Geomorphology in Urban and Forest Areas.* Wigmosta, M.S. and S.J. Burges (Ed.) Washington, DC: AGU, 17-38.

Borsuk, M.E., Stow, C.A. and Reckhow, K.H. 2004. Integrative environmental prediction using Bayesian networks: A synthesis of models describing estuarine eutrophication. *Ecological Modelling.* 173(2-3):219-239.

Bovee, K.D., Lamb, B.L., Bartholow, J.M., Stalnaker, C.B., Taylor, J., and Henriksen, J. 1998. Stream habitat analysis using the Instream Flow Incremental Methodology. *Information and Technology Report USGS/BRD-1998-0004.* U.S. Geological Survey, Biological Research Division, Washington, D.C.

Brabec, E., Schulte, S., and Richards, P.L. 2002. Impervious Surfaces and Water Quality: a Review of Current Literature and Its Implications for Watershed Planning. *Journal of Planning Literature* 16(4):499-514.

Bunn, S.E. and Arthington, A.H. 2002. Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. *Environmental Management* 30(4):492-507.

Caraco, D. 2000. "Dynamics of Urban Stream Channel Enlargement." In *The Practice of Watershed Protection*, T.R. Schueler and H.K. Holland (Ed.) Center for Watershed Protection, Ellicott City, MD.

Cassin, J., Fuerstenberg, R., Tear, L., Whiting, K., St. John, D., Murray, B., and Burkey, J. 2005. Development of Hydrological and Biological Indicators of Flow Alteration in Puget Sound Lowland Streams. King County Water and Land Resources Division. Seattle, Washington.

Center for Watershed Protection (CWP). 2002. Is Impervious Cover Still Important? Section 1: A Review of Recent Urban Stream Research. www.cwp.org/Impacts_Part1.PDF. Ellicott City, MD.

Center for Watershed Protection (CWP). 2003. Is Impervious Cover Still Important? Section 2: Impacts of Urbanization on Dosntream Receiving Waters. www.cwp.org/Impacts_Part2.PDF. Ellicott City, MD.

Chow, V.T. 1959. *Open-channel hydraulics*. McGraw-Hill, New York, NY, 680 p.

Clausen, B. and Biggs, B.J.F. 1997. Relationships Between Benthic Biota and Hydrological Indices in New Zealand Streams. *Freshwater Biology* 38(2):327-342.

Clausen, B. and Biggs, B.J.F. 2000. Flow Variables for Ecological Studies in Temperate Streams: Groupings Based on Covariance. *Journal of Hydrology* 237(3-4):184-197.

Coffman, L.S. 2003. *Discussion of an Ecosystem Functional Basis for Protecting Receiving Waters. Protection and Restoration of Streams. 139, 10. 2003/06/23/.* Philadelphia, Pennsylvania, USA, ASCE.

Coles, J.F., Cuffney, T.F., and McMahon, G., 2004. The Effects of Urbanization on the Biological, Physical, and Chemical Characteristics of Coastal New England Streams. U.S. Geological Survey Professional Paper 1695, 47 p.

Cunnane, C. 1978. Unbiased plotting positions – a review. *Journal of Hydrology*. 37:205-222.

Davis, J.P. *In preparation*. Guidelines for Rural Watershed Calibration. Master's Thesis. Colorado State University. Fall, 2007.

Dytham, C. 1999. Choosing and Using Statistics A Biologists Guide. Blackwell Science. Malden, MA. 218p.

Edgerly, J.L. 2006. "Quantifying urban-induced flow regime alteration using mathematical models and hydrologic metrics." Thesis, Department of Civil Engineering, Colorado State University, Fort Collins, CO.

Edgerly, J.L., Roesner, L.A., Rohrer, C.A., and Gironas, J.A. 2006. Quantifying Urban-induced Flow Regime Alteration and Evaluating Mitigation Alternatives Using Mathematical Models and Hydrologic Metrics. /World Environmental and Water Resources Congress: Examining the Confluence of Environmental and Water Concerns/. Omaha, NE, May 21-25.

Eisele, M., Steinbrich, A., Hildebrand, A., and Leibundgut, C. 2003. The Significance of Hydrological Criteria for the Assessment of the Ecological Quality in River Basins. *Physics and Chemistry of the Earth* 28(12-13):529-536.

Erskine, W.D., Terrazzolo, N., and Warner, R.F. 1999. River rehabilitation from the hydrologic and geomorphic impacts of a large hydro-electric power project: Snowy River, Australia. *Regul. Rivers: Res. Mgmt.* 15:3-24.

Fitzpatrick, F.A. and Peppler, M.C. 2007. Changes in aquatic habitat and geomorphic response to urbanization, with implications for assessing habitat degradation. EWRI 2007. *Restoring our Natural Habitat. Proceedings of the 2007 World Environmental and Water Resources Congress.* May 15-19, 2007. Tampa, FL.

Fitzpatrick, F.A., Waite, I.R., D'Arconte, P.J., Meador, M.R., Maupin, M.A., and Gurtz, M.E. 1998. *Revised Methods for Characterizing Stream Habitat in the National Water-Quality Assessment Program, Water-Resources Investigations Report 98-4052*, U.S. Geological Survey, Raleigh, NC.

Fitzpatrick, F.A., Diebel, M.W., Harris, M.A., Arnold, T.L., Lutz, M.A. and Richards, K.D. 2005. Effects of Urbanzaiton on the Geomorphology, Habitat, Hydrology, and Fish Index of Biotic Integrity of Streams in the Chicago Area, Illinois and Wisconsin. American Fisheries Society Symposium. 47:87-115.

Foster, G.R., Meyer, L.D. and Onstad, C.A. 1977. An erosion equation derived from basic erosion principles. Trans. ASAE 20(4):678–682.

Giddings, E.M.P. 2005. Personal Communication. Email sent to Christine Rohrer 9 Nov 2005.

Giddings, E.M.P. 2006. Personal Communication. Email sent to Christine Rohrer 19 June 2006.

Giddings, E.M.P., and Steuer, J.J. 2007. Hydrologic variability in urban streams of the North Carolina Piedmont. In Preparation.

Gore, J.A., Layzer, J.B., and Mead, J. 2001. Macroinvertebrate Instream Flow Studies After 20 Years: a Role in Stream Management and Restoration. *Regulated Rivers-Research & Management* 17(4-5):527-542.

Graf, W.L. 1975. The Impact of Suburbanization on Fluvial Geomorphology. *Water Resources Research.* 11:690-692.

Grissinger, E.H. 1982. Bank erosion of cohesive materials. In: Hey, R.D., Bathurst, J.C., and Thorne, C.R. (eds.) Gravel-bed Rivers. Wiley: Chichester, pp. 273-287.

Hammer, T.R. 1972. Stream Channel Enlargement Due to Urbanization. *Water Resources Research*. 8:1530-1540.

Hanson, G.J. and Cook, K.R. 2004. Apparatus, test procedures, and analytical methods to measure soil erodibility in situ. *Applied Engineering in Agriculture* 20(4):455-462.

Harrelson, C.C., Rawlins, C.L., Potyondy, J.P. 1994. Stream channel reference sites: an illustrated guide to field technique. Gen. Tech. Rep. RM-245. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 61 p.

Heineman, M. 2007. NETStorm Help. Available from http://www.dynsystem.com/netstorm/ Accessed 7/3/2007.

Henriksen, J.A., Heasley, J., Kennen, J.G., and Nieswand, S. 2006. User's manual for the Hydoecological Integrity Assessment Process software (including the New Jeresy Assessment Tools). Fort Collins, CO: U.S. Geological Survey, Fort Collins Science Center. Open-File Report 2006-1093. 71 p.

Hollis, G.E. 1975. Effect of urbanization on floods of different recurrence interval. *Water Resources Research* 11(3):431-435.

Holman-Dodds, J.K., Bradley, A.A., and Potter, K.W. 2003. Evaluation of Hydrologic Benefits of Infiltration Based Urban Storm Water Management. *Journal of the American Water Resources Association*. 39(1):205-215.

Horner, R., May, C.W., Livingston, E., Blaha, D., Scoggins, M., Tims, J., and Maxted, J. 2001. Structural and non-structural BMPs for protecting streams. In 'Linking Stormwater BMP Designs and Performance to Receiving Water Impact Mitigation. Engineering Research Foundation Conference, Snowmass, Colorado. (Ed. Urbonas, B.) pp.60-77. American Society of Civil Engineers: Reston, VA.

Huber, W.C. and Dickinson, R.E. 1998. *Storm Water Management Model User's Manual.* Version 4.4, EPA/600/3?88/001a, Environmental Protection Agency, Athens, GA.

Jacobson, R.B., Femmer, S.R. and McKenney, R.A. 2001.*Land-use changes and the physical habitat of streams-A review with emphasis on studies within the U.S. Geological Survey Federal-State Cooperative Program.* U.S. Geological Survey, Circular 1175, 63.

Julien, P.Y. 1998. *Erosion and Sedimentation.* Cambridge University Press. New York, New York. 280p.

Kauffman, P.R. 1993. Physical Habitat. In: Hughes, R.M. (ed.). *Stream indicator and design workshop.* pp. 59-69. EPA-600-R-93-138. US Environmental Protection Agency, Office of Research and Development, Corvallis, OR.

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

Kennen, J.G. and Ayers, M.A. 2002. Relation of Environmental Characteristics to the Composition of Aquatic Assemblages Along a Gradient of Urban Land Use in New Jersey, 1996-98. U.S. Geological Survey. Water-Resources Investigations Report 02-4069. West Trenton, NJ. 78 p.

Kerans, B.L. and Karr, J.R. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. *Ecological Applications.* 4(4):768-785.

Kirby, C.W. 2003. *Benthic macroinvertebrate response to post-development stream hydrology and hydraulics. PhD Dissertation.* George Mason Univ., Fairfax, VA.UMI Microform 3079343.

Konrad, C.P. and Booth, D.B. 2002. *Hydrologic trends resulting from urban development in western Washington streams.* U.S. Geological Survey Water-Resources Investigation Report, 02-4040. 40 pp.

Konrad, C.P., Booth, D.B., and Burges, S.J. 2005. Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns: Consequences for channel form and streambed disturbance. *Water Resources Research* 41.

Konrad, C.P. and Burges, S.J. 2001. Hydrologic Mitigation Using on-Site Residential Storm-Water Detention. *Journal of Water Resources Planning and Management-Asce* 127(2):99-107.

Kwiatkowski, B. 2004. Critical shear for Cedar Creek. Email communication from Kwiatkowski, B. to Postel, N. August 5, 2004.

Lammert, M. and Allan, J.D. 1999. Environmental Auditing, Assessing Biotic Integrity of Streams: Effects of Scale in Measuring the Influence of Land Use/Cover and Habitat Structure on Fish and Macroinvertebrates. *Environmental Management* 23(2):257-270.

Lamouroux, N., Oliver, J.M., Persat, H., Pouilly, M., Souchon, Y., and Statzner, B. 1999. Predicting community characteristics from habitat conditions: fluvial fish and hydraulics. *Freshwater Biology* 42:275-299.

Lane, E.W. 1955. The Importance of Fluvial Geomorphology in Hydraulic Engineering. Proc. 81, Paper 745:1-17.

Lee, J.G. and Heaney, J.P. 2003. Estimation of Urban Imperviousness and Its Impacts on Storm Water Systems. *Journal of Water Resources Planning and Management-Asce* 129(5):419-426.

Leopold, L.B. 1973. River channel change with time: an example. *Geological Society of America Bulletin.* 84:1845-1860

MacRae, C.R. 1997. "Experience from morphological research on Canadian streams: Is control of the two-year frequency runoff event the best basis for stream channel protection?" In *Effects of Watershed Development and Management on Aquatic Ecosystems, Proceedings of an Engineering Conference.* Roesner, L.A. (Ed.), ASCE, 144-162.

Maddock, I. 1999. The Importance of Physical Habitat Assessment for Evaluating River Health. *Freshwater Biology* 41(2):373-391.

Maddock, I., Thoms, M., Jonson, K., Dyer, F., and Lintermans, M. 2004. Identifying the Influence of Channel Morphology on Physical Habitat Availability for Native Fish: Application to the Two-Spined Blackfish (Gadopsis Bispinosus) in the Cotter River, Australia. *Marine and Freshwater Research* 55(2):173-184.

Maxted, J. and Shaver, E. 1996. "The use of retention basins to mitigate stormwater impacts on aquatic life." In *Effects of Watershed Development and Management on Aquatic Ecosystems.* Roesner, L.A. (Ed.), Snowbird, UT: American Society of Civil Engineers, 494-512.

May, C.W. and Horner, R.R. 2002. *The Limitations of Mitigation-Based Stormwater Management in the Pacific Northwest and the Potential of a Conservation Strategy Based on Low-Impact Development Principles.* Urban Drainage Portland, OR: Sep 8-:16. ASCE;

McCuen, R.H. and Moglen, G.E. 1988. Multicriterion stormwater management methods. *Journal of Water Resources Planning and Management* 144(4):414-431.

McMahon, G., Bales, J.D., Coles, J.F., Giddings, E.M.P., and Zappia, H. 2003. Use of Stage Data to Characterize Hydrologic Conditions in an Urbanizing Environment. *Journal of the American Water Resources Association* 39(6):1529-1546.

McMahon, G. and Cuffney, T.F. 2000. Quantifying Urban Intensity in Drainage Basins for Assessing Stream Ecological Conditions. Journal of the American Water Resources Association (JAWRA) 36:1247-1261.

Merigoux, S. and Doledec, S. 2004. Hydraulic Requirements of Stream Communities: a Case Study on Invertebrates. *Freshwater Biology* 49(5):600-613.

Meyer, J.L., Paul, M.J., and.Taulbee, W.K . 2005. Stream ecosystem function in urbanizing landscapes. *J. N. Am. Benthol. Soc.* 24(3):602–612.

Mid-America Regional Council (MARC) and American Public Works Association (APWA). 2003. Manual for Best Management Practices for Stormwater Quality.

Milhous, R.T., Wegner, D.L., Waddle, T. 1984. *User's Guide to the Physical Habitat Simulation System*. US Fish and Wildlife Service, FWS/OBS 81/43: Washington, D.C.

Moglen, G.E. and R.H. McCuen. 1988. Effects of Detention Basins on In-Stream Sediment Movement. *Journal of Hydrology*. 104:129-139.

Moulton, S.R., II, Kennen, J.G., Goldstein, R.M., and Hambrook, J.A., 2002. Revised Protocols for Sampling Algal, Invertebrate, and Fish Communities as Part of the National Water-Quality Assessment Program. U.S. Geological Survey Open-File Report 02-150, 75 p.

Nehrke, S.M. and Roesner, L.A., 2004. Effect of Drainage Controls and BMPs on the Flow Frequency Curve of Urban Runoff. *Journal of Water Resources Planning and Management.* 130(22):131-139.

Neller, R. J. 1988. A Comparison of Channel Erosion in Small Urban and Rural Catchments, Armidale, New South Wales. *Earth Surfaces Processes and Landforms*. 13:1-7.

Neller, R. J. 1989. Induced Channel Enlargement in Small Urban Catchments , Armidale, New South Wales. *Environ. Geol Water Sci.* 14:167-171.

Nilsson, C. and Svedmark, M. 2002. Basic Principles and Ecological Consequences of Changing Water Regimes: Riparian Plant Communities. *Environmental Management* 30(4):468-480.

The Nature Conservancy. 2007. Indicators of Hydrologic Alteration Version 7 User's Manual.

North Carolina Department of Environment and Natural Resources (NCDENR). 2006. Standard Operating Procedures for Benthic Macroinvertebrates, Biological Assessment Unit, July 2006. North Carolina Department of Natural Resources, Division of Water Quality, Environmental Sciences Section.

North Carolina Department of Transportation (NCDOT) Geographic Information Systems (GIS). 2005. GIS Contours and Elevation Data. Accessed 11 March 2005. http://www.ncdot.org/it/gis/DataDistribution/ContourElevationData/.

Olden, J.D. and Poff, N.L. 2003. Redundancy and the Choice of Hydrologic Indices for Characterizing Streamflow Regimes. *River Research and Applications* 19(2):101-121.

Osmundson, D.B., Ryel, R.J., Lamarra, V.L., and Pitlick, J. 2002. Flow-Sediment-Biota Relations: Implications for River Regulation Effects on Native Fish Abundance. *Ecological Applications* 12(6):1719-1739.

Ott, R.L. and Longnecker, M.T. 2000. *An Introduction to Statistical Methods and Data Analysis*. Duxbury Press. 1184 p.

Parker, G. In Press. Transport of gravel and sediment mixtures, *ASCE Manual 54, Sediment Enginering*, ASCE, Chapter 3, downloadable from http://cee.uiuc.edu/people/parkerg/manual_54.htm .

Paulson, C., Reeve,. S., Canton, S.P., and Moore, T.F. 2001. Distinguishing The Relative Influence Of Habitat And Water Quality On Aquatic Biota. Final Report PROJECT 98-WSM-1 to Water Environment Research Foundation.

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E., and Stromberg, J.C. 1997. The Natural Flow Regime. *Bioscience* 47(11):769-784.

Poff, N.L. 1996. A Hydrogeography of Unregulated Streams in the United States and an Examination of Scale-dependence in Some Hydrological Descriptors. *Freshwater Biology* 36:71-91.

Poff, N.L., and Ward, J.V. 1989. Implications of Streamflow Variability and Predictability for Lotic Community Structure - a Regional-Analysis of Streamflow Patterns. *Canadian Journal of Fisheries and Aquatic Sciences* 46(10):1805-1818.

Pomeroy, C.A., N.A. Postel, P.A. O'Neill, and L.A. Roesner. *In press a*. Development of Stormwater Management Design Criteria to Maintain Geomorphic Stability in a Kansas City Metropolitan Area Stream. Accepted by Journal of Irrigation and Drainage.

Pomeroy, C.A., L.A. Roesner and J.C. Coleman. *In press b*. Protocols for Evaluating Wet Weather Practices and Urbanization Patterns. Final report to Water Environment Research Foundation. WERF, Alexandria, Virginia. 03-WSM-03.

Pomeroy, C.A. and L.A. Roesner. *In press*. Tools for the evaluation of stormwater management practices that provide ecological stability in urban streams. In *Cities of the Future: Towards Integrated Sustainable Water and Landscape Management*, (V. *Management* Novotny, L. Breckenridge, and P. Brown, Eds.), IWA Publishing, London UK.

Rankin, E.T. 1989. *The Qualitative Habitat Evaluation Index (Qhei), Rational, Methods, and Application,* Ohio EPA, Columbus, OH.

Rankin, E.T. 1995. *The Use of habitat Assessments in Water Resource Programs, Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making,* Lewis Publishers, Boca Raton, FL.

Reed, T.J., Centinaro, G.L., DeLuca, M.J., Hutchinson, J.T., and Scudder, J. 1997. Water resources data for New Jersey –Water year 1996, Volume 1. Surface-water data: U.S. Geological Survey Water Data Report, NJ-96-1, 562 p.

Richter, B.D., Baumgartner, J.V., Powell, J., and Braun, D.P. 1996. A method for assessing hydrologic alteration within ecosystems. *Conservation Biology* 10:1163-1174.

Richter, B.D., Baumgartner, J.V., Braun, D.P., and Powell, J. 1998. A Spatial Assessment of Hydrologic Alteration Within a River Network. *Regulated Rivers-Research & Management* 14 (4):329-340.

Richter, B.D., Baumgartner, J.V., Wigington, R., and Braun, D.P. 1997. How Much Water Does a River Need? *Freshwater Biology* 37(1):231-249.

Richter, B.D., Mathews, R., and Wigington, R. 2003. Ecologically Sustainable Water Management: Managing River Flows for Ecological Integrity. *Ecological Applications* 13(1): 206-224.

Richter, B.D. and Richter, H.E. 2000. Prescribing Flood Regimes to Sustain Riparian Ecosystems Along Meandering Rivers. *Conservation Biology* 14(5):1467-1478.

Riis, T. and Biggs, B.J.F. 2003. Hydrologic and Hydraulic Control of Macrophyte Establishment and Performance in Streams. *Limnology and Oceanography* 48(4):1488-1497.

Roesner, L.A. and Bledsoe, B.P. 2003. *Research Needs: Physical Effects of Wet Weather Flows on Aquatic Habitats.* WERF. 00-WSM-04.

Rohrer, C.A. 2004. "Modeling the Effect of Stormwater Controls on Sediment Transport in an Urban Stream." Thesis, Department of Civil Engineering, Colorado State University, Fort Collins, CO.

Rohrer, C.A., Postel, N.A., O'Neill, P.A., and Roesner, L.A. 2005. Development of Design Criteria for Regional Stormwater Management Facilities to Maintain Geomorphic Stability in Cedar Creek. World Water & Environmental Resources Congress: Impacts of Global Climate Change on Water Resources and the Environment. Anchorage, AK, May 15-19.

Rohrer, C.A. and Roesner, L.A. 2006. Matching the Critical Portion of the Flow Duration Curve to Minimize Changes in Modeled Excess Shear. Water Science and Technology. 54:347-354.

Rossman, L.A. 2005. Storm Water Management Model User's Manual. Version 5, EPA/600/R-05/040, Environmental Protection Agency, Athens, GA.

Scoggins, M. 2000. *Effects of Hydrologic Variablility on Biological Assessments in Streams in Austin, TX.* City of Austin, Watershed Protection Department. NWQMC 2000 proceedings.

Shaver, E. and Ridley, G.. 2002. *Our Use of the Land: Troubling Times and Changing Directions. Urban Drainage Portland, Oregon,* USA: ASCE; Sep 8-: 255.

Shields, A. 1936. Anwendung der Aehnlichkeitsmechanik und der Turbulenz Forschung auf die Geschiebebewegung. Berlin: Mitteilungun der Preussische Versuchanstalt für Wasserbau und Schiffbau.

State Climate Office of North Carolina. 2007. http://www.nc_climate.ncsu.edu/climate/hurrucane.php. Accessed 28 June 2007.

Statzner, B. and Moss, B. 2004. Linking Ecological Function, Biodiversity and Habitat: a Mini-Review Focusing on Older Ecological Literature. *Basic and Applied Ecology* 5(2):97-106.

Stewardson, M.J. and Gippel, C.J. 2003. Incorporating Flow Variability Into Environmental Flow Regimes Using the Flow Events Method. *River Res. and Applications* 19(5-6):459-472.

Suen, J.P., Herricks, E.E., and Eheart, J.W. 2004. *Ecohydrologic Indicators for Rivers of Northern Taiwan. World Water Congress 2004.* 138, 143. 2004/06/27/. Salt Lake City, UT, USA, ASCE.

Tate, C.M., Cuffney, T.F., McMahon, G., Giddings, E.M.P., and Zappia, H., 2005. Use of an Urban Intensity Index to Assess Urban Effects on Streams in Three Contrasting Environmental Settings. *In* Effects of urbanization on stream ecosystems Brown, L.R., Gray, R.H., Hughes, R.M., and Meador, M.R. (Editors). American Fisheries Society Bethesda, MD.

Tharme, R.E. 2003. A Global Perspective on Environmental Flow Assessment: Emerging Trends in the Development and Application of Environmental Flow Methodologies for Rivers. *River Research and Applications* 19(5-6):397-441.

Thomson, J.R., Taylor, M.P., and Brierley, G.J. 2004. Are River Styles Ecologically Meaningful? A Test of the Ecological Significance of a Geomorphic River Characterization Scheme. *Aquatic Conservation-Marine and Freshwater Ecosystems* 14(1):25-48.

Townsend, C.R., Downes, B.J., Peacock, K., and Arbuckle, C.J. 2004. Scale and the Detection of Land-Use Effects on Morphology, Vegetation and Macroinvertebrate Communities of Grassland Streams. *Freshwater Biology* 49(4):448-462.

Urbonas, B. 1980. Drainageway Erosion in Semi-Arid Urbanizing Areas. *Flood Hazard News*. 10:1:1-2.

U.S. EPA. 2005 Draft. *Use of Biological Information to Better Define Designated Aquatic Life Uses in State and Tribal Water Quality Standards: Tiered Aquatic Life Uses. Tiered Aquatic Life Uses Document.* EPA-822-R-05-001. U.S. Environmental Protection Agency, Office of Water.

U.S. EPA. 2006. Stormwater Management Model (SWMM5) Software Help. http://www.epa.gov/ednnrmrl/models/swmm/index.htm. Website accessed 8/2/2007.

U.S. Geological Survey (USGS). 2003. National Land Cover Database Zone 60 Imperviousness Layer. Edition 1.0. U.S. Geological Survey. Sioux Falls, SD.

U.S. Geological Survey (USGS). 2006a. National Elevation Dataset (NED) 1/3 Arc Second (~10m) Digital Elevation Map. U.S. Geological Survey. Sioux Falls, SD.

U.S. Geological Survey (USGS). 2006b. Raleigh Durham (Mar 2002) Orthoimagry. U.S. Geological Survey. Sioux Falls, S.D.

U.S. Geological Survey (USGS). 2007. National Water-Quality Assessment Program - Effects of Urbanization on Stream Ecosystems. http://co.water.usgs.gov/nawqa/EUSE/ Web site accessed 2/20/07.

U.S. National Resources Conservation Service (USNRCS). 1994. SSURGO and STATSGO data accessed from the Geospatial Data Gateway. http://datagateway.nrcs.usda.gov. Web site accessed 8/2/2005.

Wake County North Carolina. Tax/Parcel, Digital Orthophotos, Topographic & Soils Maps. 1999. Accessed 11 March 2005. http://lnweb02.co.wake.nc.us/gis/gismaps.nsf

Walsh, C.J. 2004. Protection of in-Stream Biota From Urban Impacts: Minimise Catchment Imperviousness or Improve Drainage Design? *Marine and Freshwater Research* 55(3):317-326.

Walsh, C.J., Papas, P.J., Crowther, D., Sim, P.T., and Yoo, J. 2004. Stormwater drainage pipes as a threat to a stream-dwelling amphipod of conservation significance, *Austrogammarus australis*, in south-eastern Australia. *Biodiversity and Conservation,* 13:781-793.

Walsh, C.J., Roy, A.H., Feminella, J.W., Cottingham, P.D., Groffman, P.M. and R.P. Morgan II. 2005. The urban stream syndrome: current knowledge and the search for a cure. *J. N. Am. Benthol. Soc.* 24(3):706–723.

Ward, J.V., Malard, F., and Tockner, K. 2002. Landscape ecology: a framework for integrating pattern and process in river corridors. *Landscape Ecology* 17(Suppl. 1): 34-45.

Water Environment Federation (WEF) and American Society of Civil Engineers (ASCE). 1998. *Urban Runoff Quality Management.* WEF manual of practice; no. 23. ASCE manual and report on engineering practice 87.

Waters, T.F. 1995. Sediment in streams - Sources, biological effects and control. *American Fisheries Society Monograph 7.* American Fisheries Society, Bethesda, MD.

Whipple, W. Jr., J.M. DiLouie and T. Pytlar, Jr. 1981. Erosional Potential of Streams in Urbanizing Areas. *Water Resources Bulletin.* 17:36-45.

Winward, A.H. 2000. *Monitoring the vegetation resources in riparian areas.* Gen. Tech. Rep. RMRSGTR-47. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 49 pp.

Wolman, M. G. 1967. A Cycle of Sedimentation and Erosion in Urban River Channels. *Geografiska Annaler*. 49A:385-394.

Wu, F.C. and Wang, C.F. 2002. Effect of Flow-Related Substrate Alteration on Physical Habitat: a Case Study of the Endemic River Loach Sinogastromyzon Puliensis (Cypriniformes, Homalopteridae) Downstream of Chi-Chi Diversion Weir, Chou-Shui Creek, Taiwan. *River Research and Applications* 18(2):155-169.

Wulliman, J. and Urbonas, B. 2005. Peak Flow Control for Full Spectrum of Design Storms. Urban Drainage and Flood Control District Concept Paper.

Yoder, C.O. 1998. Important concepts and elements of an adequate state watershed monitoring and assessment program. Pages 615–628 in: *Proceedings of the NWQMC National Conference on Monitoring: Critical Foundations to Protecting Our Waters*. U.S. Environmental Protection Agency, Washington, D.C.

APPENDIX

Table A-1. Summary of Recommended Hydrologic Indices That Best Represent the Biologically Critical Portions of the Flow Regime, by Stream Type. (After Olden and Poff, 2003).

		Definition	Intermittent: Harsh Intermittent	Intermittent: Intermittent Flashy or Runoff	Perennial: Snowmelt	Perennial: Snow & Rain	Perennial: Superstable or Stable Groundwater	Perennial: Perennial Flashy or Runoff	All Streams
Magnitude of Flow Events: Average Flow Conditions									
MA3	Variability in daily flows 1	Coefficient of variation in daily flows				x	x		x
MA5	Skewness in daily flows	Mean daily flows divided by median daily flows							x
MA8	Ranges in daily flows	Ratio of 10th/90th, 20th/80th and 25th/75th percentiles in daily flows over all years					x		
MA9	Spreads in daily flows	Ranges in daily flows (MA6–8) divided by median daily flows		x					
MA10	Spreads in daily flows	Ranges in daily flows (MA6–8) divided by median daily flows						x	
MA11	Spreads in daily flows	Ranges in daily flows (MA6–8) divided by median daily flows							x
MA16	Mean monthly flows	Mean monthly flow for all months	x						
MA18	Mean monthly flows	Mean monthly flow for all months		x					
MA21	Mean monthly flows	Mean monthly flow for all months		x					
MA22	Mean monthly flows	Mean monthly flow for all months	x						
MA26	Variability in monthly flows	Coefficient of variation in monthly flows for all months						x	
MA29	Variability in monthly flows	Coefficient of variation in monthly flows for all months			x				
MA34	Variability in monthly flows	Coefficient of variation in monthly flows for all months	x						
MA37	Variability across monthly flows 1	Variability in monthly flows divided by median monthly flows, where variability is calculated as range, interquartile, and 90th–10th percentile.		x					
MA40	Skewness in monthly flows	(Mean monthly flow—median monthly flow)/median monthly flow			x				

			Intermittent			Perennial			
		Definition	Harsh Intermittent	Intermittent Flashy or Runoff	Snowmelt	Snow & Rain	Superstable or Stable Groundwater	Perennial Flashy or Runoff	All Streams
M_A41	Mean annual runoff	Mean annual flow divided by catchment area					x	x	x
M_A44	Variability across annual flows	Variability in annual flows divided by median annual flows, where variability is calculated as range, interquartile, and 90th–10th percentile.				x			
Magnitude of Flow Events: Low Flow Conditions									
M_L1	Mean minimum monthly flows	Mean minimum monthly flow for all months	x						
M_L4	Mean minimum monthly flows	Mean minimum monthly flow for all months							x
M_L6	Mean minimum monthly flows	Mean minimum monthly flow for all months		x					
M_L13	Variability across minimum monthly flows	Coefficient of variation in minimum monthly flows	x		x	x			
M_L14	Mean of annual minimum flows	Mean of the lowest annual daily flow divided by median annual daily flow averaged across all years				x	x	x	
M_L15	Low flow index	Mean of the lowest annual daily flow divided by mean annual daily flow averaged across all years	x	X					
M_L16	Median of annual minimum flows 2	Median of the lowest annual daily flows divided by median annual daily flows averaged across all years		x			x	x	
M_L17	Baseflow index 1	Seven-day minimum flow divided by mean annual daily flows averaged across all years						x	x
M_L18	Variability in Baseflow Index 1	Coefficient of variation in M_L17							x
M_L21	Variability across annual minimum flows	Coefficient of variation in annual minimum flows averaged across all years							x
M_L22	Specific mean annual minimum flows	Mean annual minimum flows divided by catchment area		x	x				
Magnitude of Flow Events: High Flow Conditions									
M_H1	Mean maximum monthly flows	Mean of the maximum monthly flows for all months			x				
M_H4	Mean maximum monthly flows	Mean of the maximum monthly flows for all months		x					

		Definition	Intermittent		Perennial				
			Harsh Intermittent	Intermittent Flashy or Runoff	Snowmelt	Snow & Rain	Superstable or Stable Groundwater	Perennial Flashy or Runoff	All Streams
$M_{H}7$	Mean maximum monthly flows	Mean of the maximum monthly flows for all months		x					
$M_{H}8$	Mean maximum monthly flows	Mean of the maximum monthly flows for all months						x	x
$M_{H}9$	Mean maximum monthly flows	Mean of the maximum monthly flows for all months	x						
$M_{H}10$	Mean maximum monthly flows	Mean of the maximum monthly flows for all months					x		x
$M_{H}14$	Median of annual maximum flows	Median of the highest annual daily flow divided by the median annual daily flow averaged across all years	x	x				x	x
$M_{H}16$	High flow discharge	Mean of the 1st, 10th and 25th percentile from the flow duration curve divided by median daily flow across all years							x
$M_{H}17$	High flow discharge	Mean of the 1st, 10th and 25th percentile from the flow duration curve divided by median daily flow across all years				x	x		
$M_{H}19$	Skewness in annual maximum flows	See Hughes and James (1989)					x		
$M_{H}20$	Specific mean annual maximum flows	Mean annual maximum flows divided by catchment area			x	x			
$M_{H}23$	High flow volume	Mean of the high flow volume (calculated as the area between the hydrograph and the upper threshold during the high flow event) divided by median annual daily flow across all years. The upper threshold is defined as 1, 3, and 7 times median annual flow	x	x				x	
Frequency of Flow Events: Low Flow Conditions									
$F_{L}1$	Low flood pulse count	Number of annual occurrences during which the magnitude of flow remains below a lower threshold. Hydrologic pulses are defined as those periods within a year in which the flow drops below the 25th percentile (low pulse) of all daily values for the time period	x	x			x		x
$F_{L}2$	Variability in low Flood pulse count	Coefficient of variation in $F_{L}1$	x	x	x	x	x	x	x
$F_{L}3$	Frequency of low flow spells	Total number of low flow spells (threshold equal to 5% of mean daily flow) divided by the record length in years	x	x	x	x	x	x	x

			Intermittent		Perennial				
		Definition	Harsh Intermittent	Intermittent Flashy or Runoff	Snowmelt	Snow & Rain	Superstable or Stable Groundwater	Perennial Flashy or Runoff	All Streams
Frequency of Flow Events: High Flow Conditions									
F_H2	Variability in high flood pulse count 1	Coefficient of variation in F_H1	x	x					x
F_H3	High flood pulse count 2	See F_H1, where the upper threshold is defined as 3 and 7 times median daily flow, and the value is represented as an average instead of a tabulated count		x		x	x		x
F_H4	High flood pulse count 2	See F_H1, where the upper threshold is defined as 3 and 7 times median daily flow, and the value is represented as an average instead of a tabulated count						x	
F_H5	Flood frequency 1	Mean number of high flow events per year using an upper threshold of 1, 3, and 7 times median flow over all years	x			x			
F_H6	Flood frequency 1	Mean number of high flow events per year using an upper threshold of 1, 3, and 7 times median flow over all years					x	x	x
F_H7	Flood frequency 1	Mean number of high flow events per year using an upper threshold of 1, 3, and 7 times median flow over all years	x	x				x	x
F_H8	Flood frequency 2	See F_H5–7, where the 25th and 75th percentile are used as the upper threshold			x				
F_H10	Flood frequency 3	See F_H5–7, where the median of the annual minima is used as the upper threshold		x					
F_H11	Flood frequency 4	Mean number of discrete flood events per year			x		x		
Duration of Flow Events: Low Flow Conditions									
D_L1	Annual minima of 1-/3-/7-/30-/90-day means of daily discharge	Magnitude of minimum annual flow of various duration, ranging from daily to seasonal	x	x					
D_L2	Annual minima of 1-/3-/7-/30-/90-day means of daily discharge	Magnitude of minimum annual flow of various duration, ranging from daily to seasonal	x						
D_L5	Annual minima of 1-/3-/7-/30-/90-day means of daily discharge	Magnitude of minimum annual flow of various duration, ranging from daily to seasonal			x				
D_L6	Variability in annual minima of one-/3-/7-/30-/90-day means of daily discharge	Coefficient of variation in D_L1-5				x		x	
D_L9	Variability in annual minima of one-/3-/7-/30-/90-day means of daily discharge	Coefficient of variation in D_L1-5					x		

			Intermittent			Perennial			
		Definition	Harsh Intermittent	Intermittent Flashy or Runoff	Snowmelt	Snow & Rain	Superstable or Stable Groundwater	Perennial Flashy or Runoff	All Streams
D_L10	Variability in annual minima of one-/3-/7-/30-/90-day means of daily discharge	Coefficient of variation in D_L1-5						x	
D_L11	Means of 1-/7-/30-/90-day minima of daily discharge	Mean annual 1-day/7-day/30-day minimum, respectively, divided by median flow					x		
D_L13	Means of 1-/7-/30-/90-day minima of daily discharge	Mean annual 1-day/7-day/30-day minimum, respectively, divided by median flow	x	x		x			x
D_L16	Low flow pulse duration	Mean duration of F_L1		x	x		x		x
D_L17	Variability in low flow pulse duration	Coefficient of variation in D_L16						x	x
D_L18	Number of zero-flow days	Mean annual number of days having zero daily flow		x					x
Duration of Flow Events: High Flow Conditions									
D_H5	Annual maxima of 1-/3-/7-/30-/90-day means of daily discharge	Magnitude of maximum annual flow of various duration, ranging from daily to seasonal	x						
D_H10	Variability in annual maxima of 1-/3-/7-/30-day means of daily discharge	Coefficient of variation in D_H1-5	x						
D_H11	Means of 1-/7-/30-day maxima of daily discharge	Mean annual 1-day/7-day/30-day maximum, respectively, divided by median flow					x		
D_H12	Means of 1-/7-/30-day maxima of daily discharge	Mean annual 1-day/7-day/30-day maximum, respectively, divided by median flow		x		x			
D_H13	Means of 1-/7-/30-day maxima of daily discharge	Mean annual 1-day/7-day/30-day maximum, respectively, divided by median flow		x				x	x
D_H15	High flow pulse duration	Mean duration of F_H1		x			x		x
D_H16	Variability in high flow pulse duration	Coefficient of variation in D_H15			x			x	x
D_H19	High flow duration 1	See DH15, where the upper threshold is defined as 1, 3, and 7 times median flows, and the value is represented as an average instead of a tabulated count			x				
D_H20	High flow duration 2	See DH17–19, where the upper threshold is defined as the 25th and 75th percentile of median flows					x		x

			Intermittent		Perennial				
		Definition	Harsh Intermittent	Intermittent Flashy or Runoff	Snowmelt	Snow & Rain	Superstable or Stable Groundwater	Perennial Flashy or Runoff	All Streams
D_H22	Flood interval	Mean annual median interval in days between floods over all years	x						
D_H23	Flood duration 2	Mean annual number of days that flows remain above the flood threshold averaged across all years		x					
D_H24	Flood free days	Mean annual maximum number of 365 days over all water years during which no floods occurred over all years				x		x	
Timing of Flow Events									
T_A1	Constancy	See Colwell (1974)		x	x	x	x	x	x
T_A2	Predictability of flow	Composed of two independent, additive components: constancy (a measure of temporal invariance) and contingency (a measure of periodicity)		x					
T_A3	Seasonal predictability of flooding	Maximum proportion of all floods over the period of record that fall in any one of six 60-day 'seasonal' windows			x			x	
T_L1	Julian date of annual minimum	The mean Julian date of the 1-day annual minimum flow over all years		x		x			
T_L2	Variability in Julian date of annual minimum	Coefficient of variation in T_L1	x				x		x
T_H1	Julian date of annual maximum	The mean Julian date of the 1-day annual maximum flow over all years	x				x		
T_H2	Variability in Julian date of annual maximum	Coefficient of variation in T_H1	x						
T_H3	Seasonal predictability of non-flooding	Maximum proportion of the year (number of days/365) during which no floods have ever occurred over the period of record		x				x	x
Rate of Change in Flow Events									
R_A1	Rise rate	Mean rate of positive changes in flow from one day to the next	x		x				
R_A4	Variability in fall rate	Coefficient of variation in RA_3	x						
R_A5	No day rises	Ratio of days where the flow is higher than the previous day	x	x			x		x
R_A6	Change of flow	Median of difference between natural logarithm of flows between two consecutive days with increasing/decreasing flow		x				x	x

			Intermittent		Perennial				
		Definition	Harsh Intermittent	Intermittent Flashy or Runoff	Snowmelt	Snow & Rain	Superstable or Stable Groundwater	Perennial Flashy or Runoff	All Streams
R_A7	Change of flow	Median of difference between natural logarithm of flows between two consecutive days with increasing/decreasing flow		x				x	
R_A8	Reversals	Number of negative and positive changes in water conditions from one day to the next			x	x	x		x
R_A9	Variability in reversals	Coefficient of variation in RA_8		x		x	x	x	x
R_A1	Rise Rate	Mean rate of positive changes in flow from one day to the next	x		x				
R_A4	Variability in fall rate	Coefficient of variation in RA_3	x						
R_A5	No day rises	Ratio of days where the flow is higher than the previous day	x	x			x		x

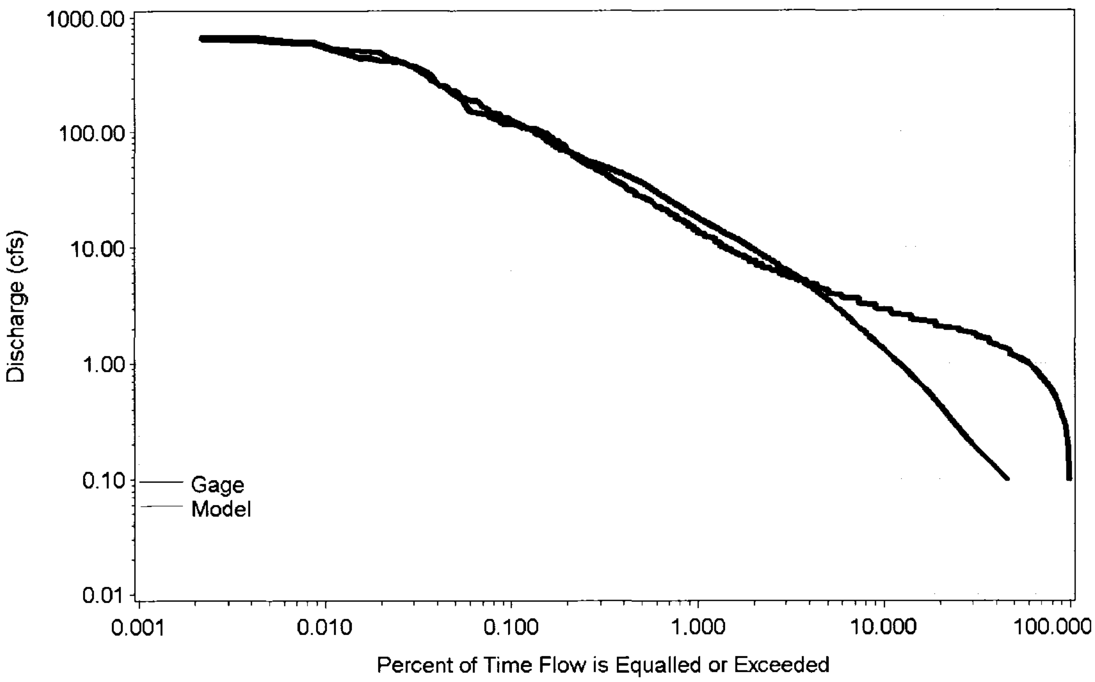


Figure A-1. Flow Duration Curve For Calibration Period: Camp Creek.

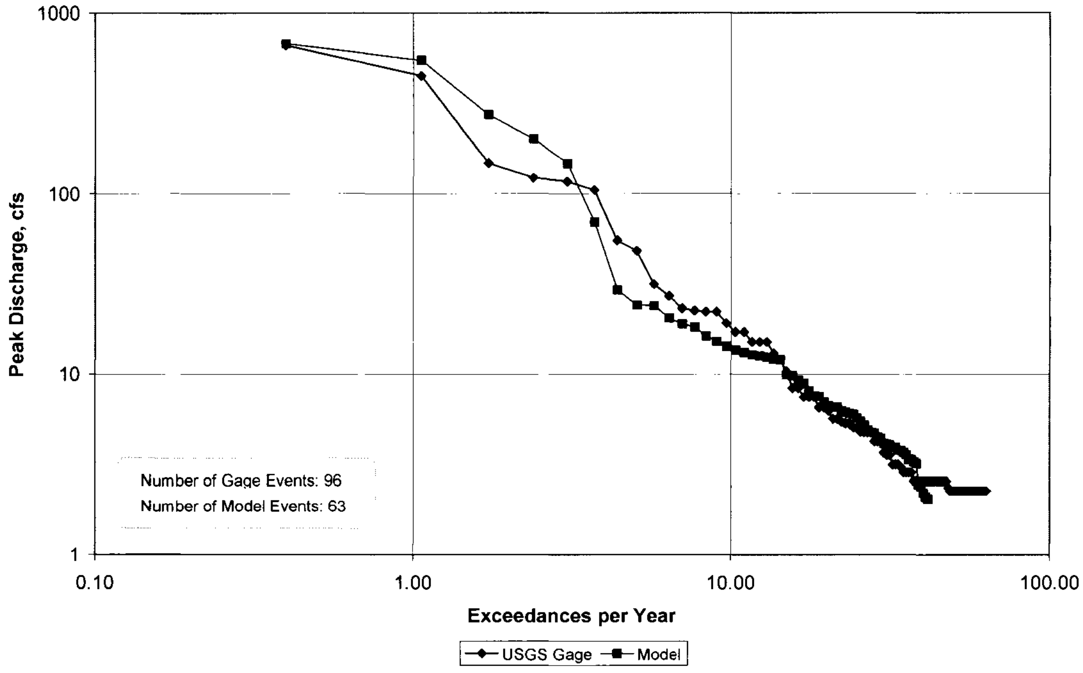


Figure A-2. Peak Flow Frequency Exceedance Curve For Calibration Period: Camp Creek.

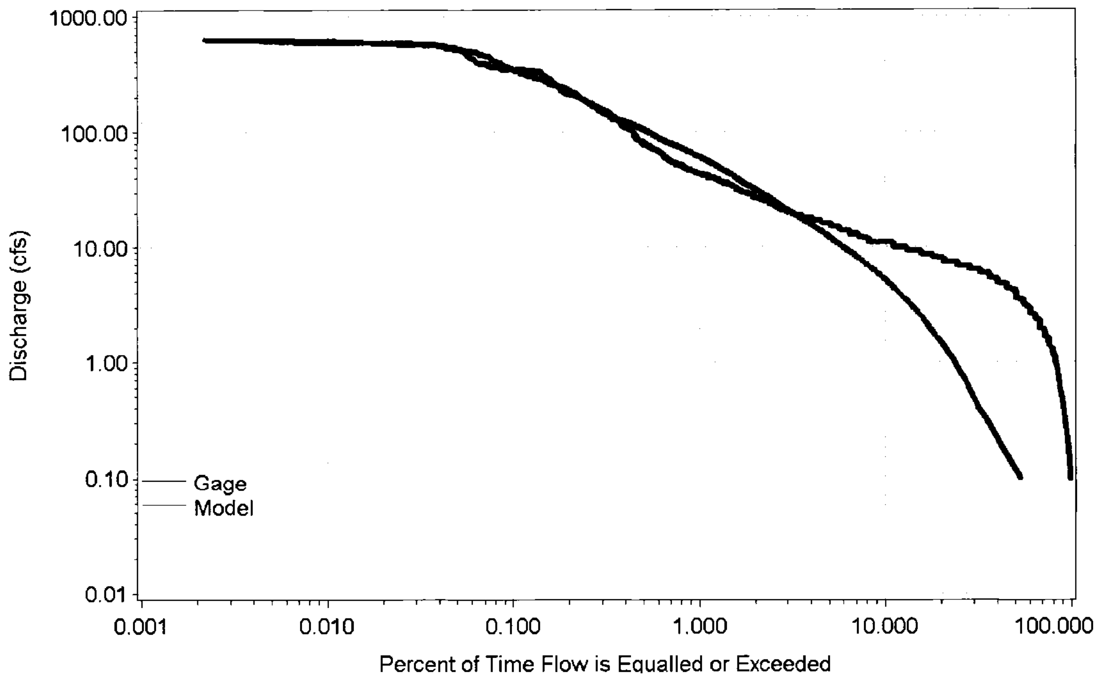


Figure A-3. Flow Duration Curve For Calibration Period: Wilson Creek.

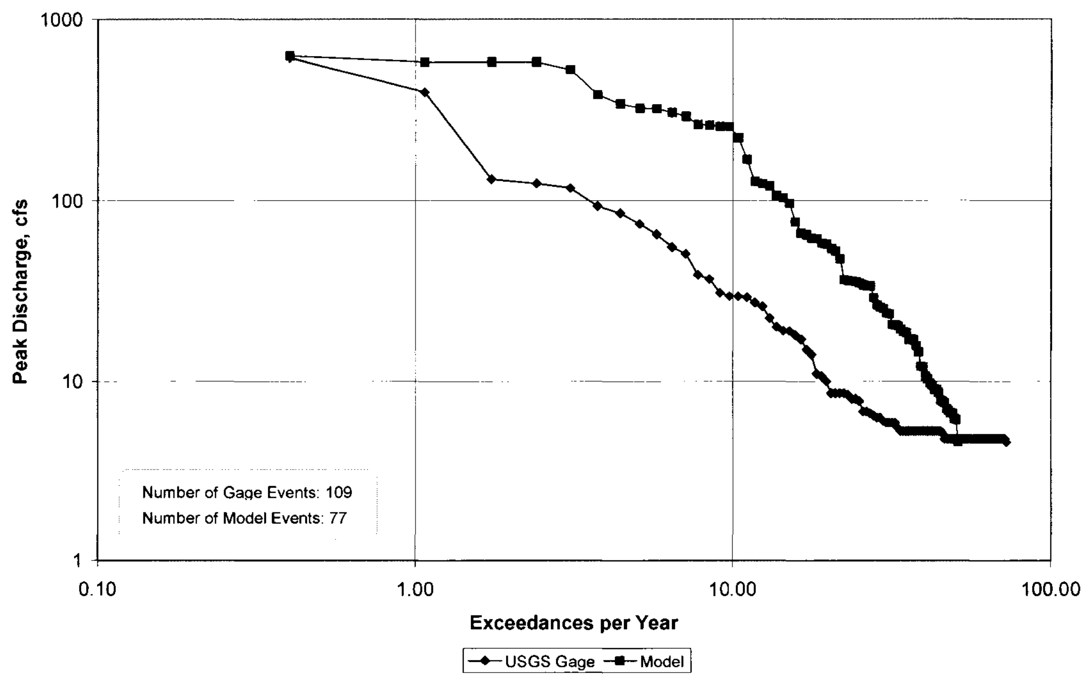


Figure A-4. Peak Flow Frequency Exceedance Curve For Calibration Period: Wilson Creek.

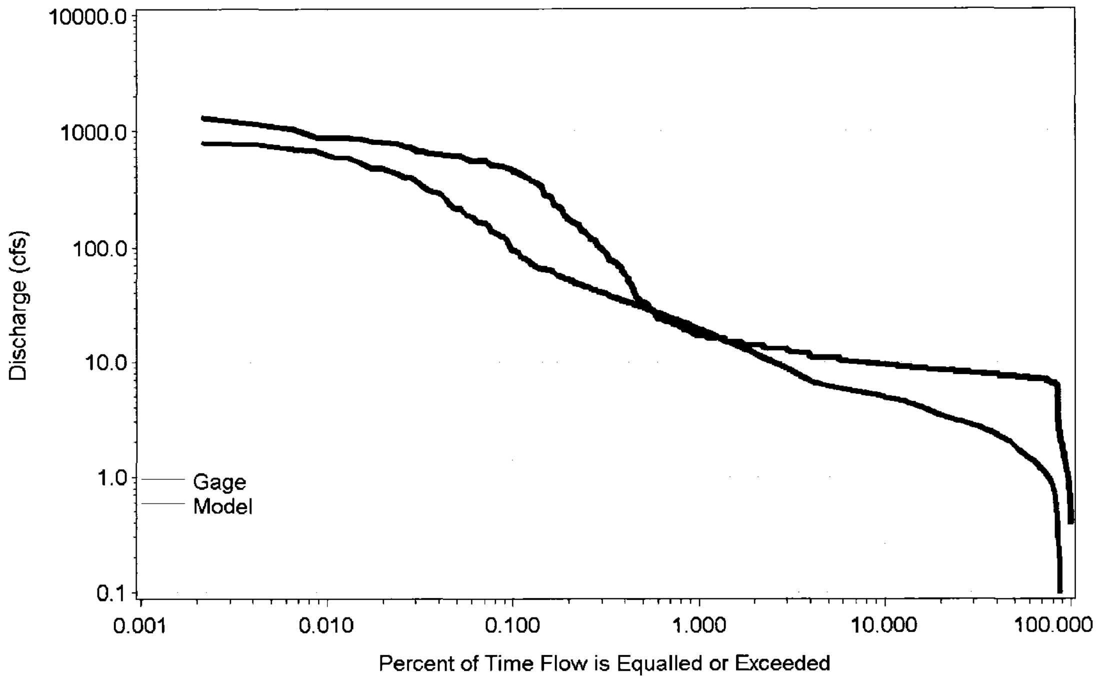


Figure A-5. Flow Duration Curve For Calibration Period: Dutchmans Creek.

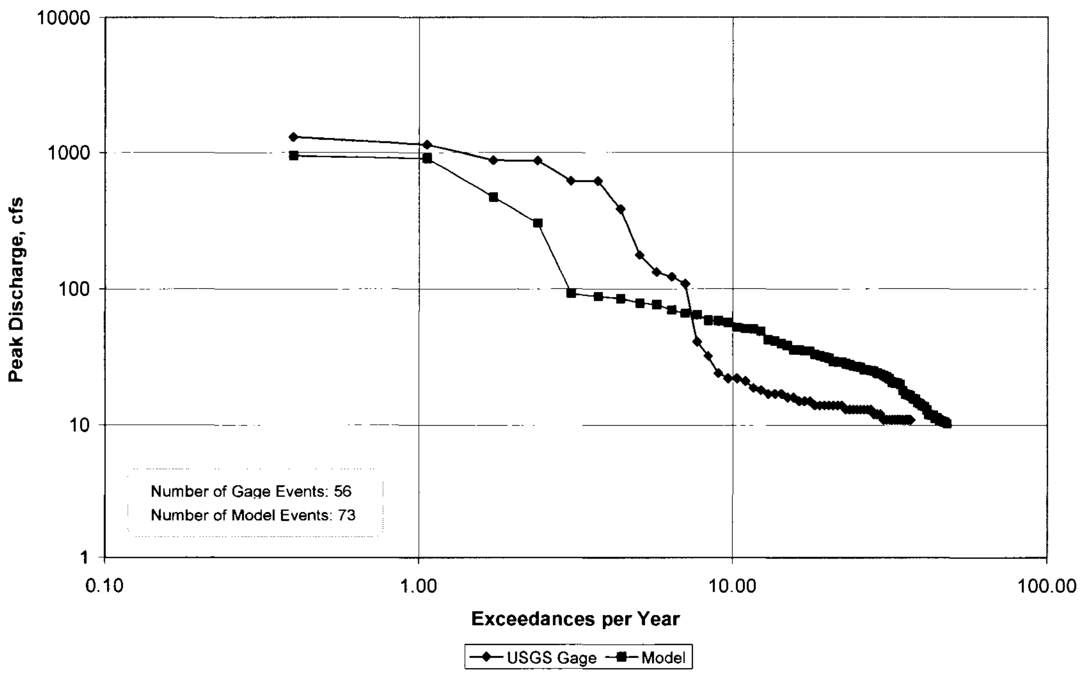


Figure A-6. Peak Flow Frequency Exceedance Curve For Calibration Period: Dutchmans Creek.

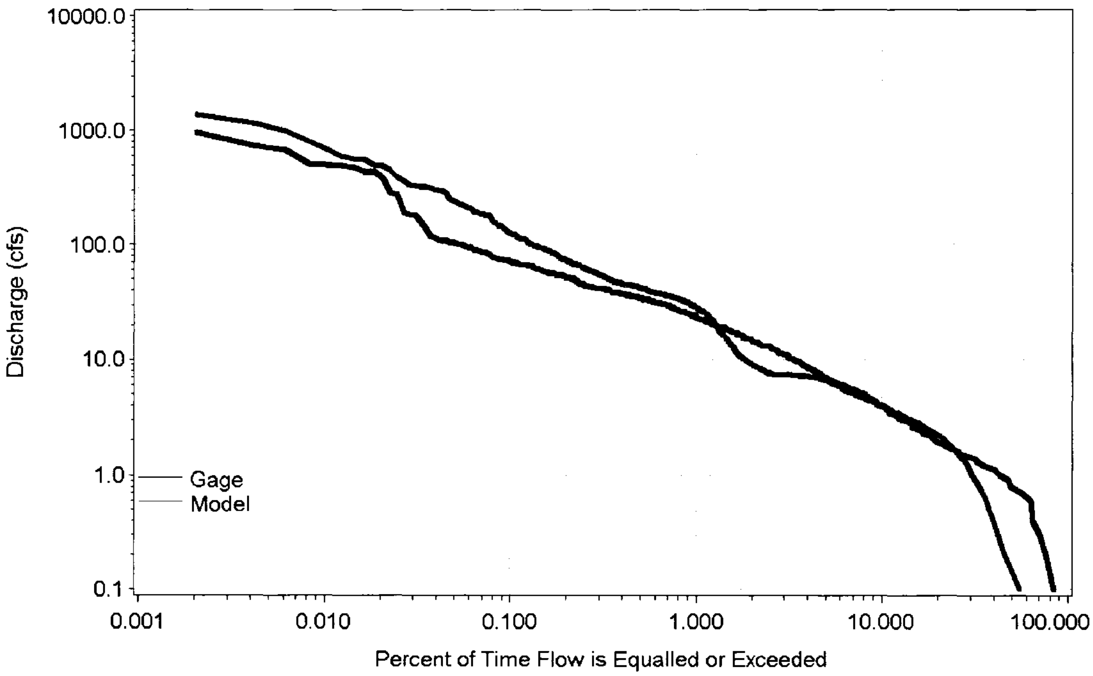


Figure A-7. Flow Duration Curve For Calibration Period: Black Creek.

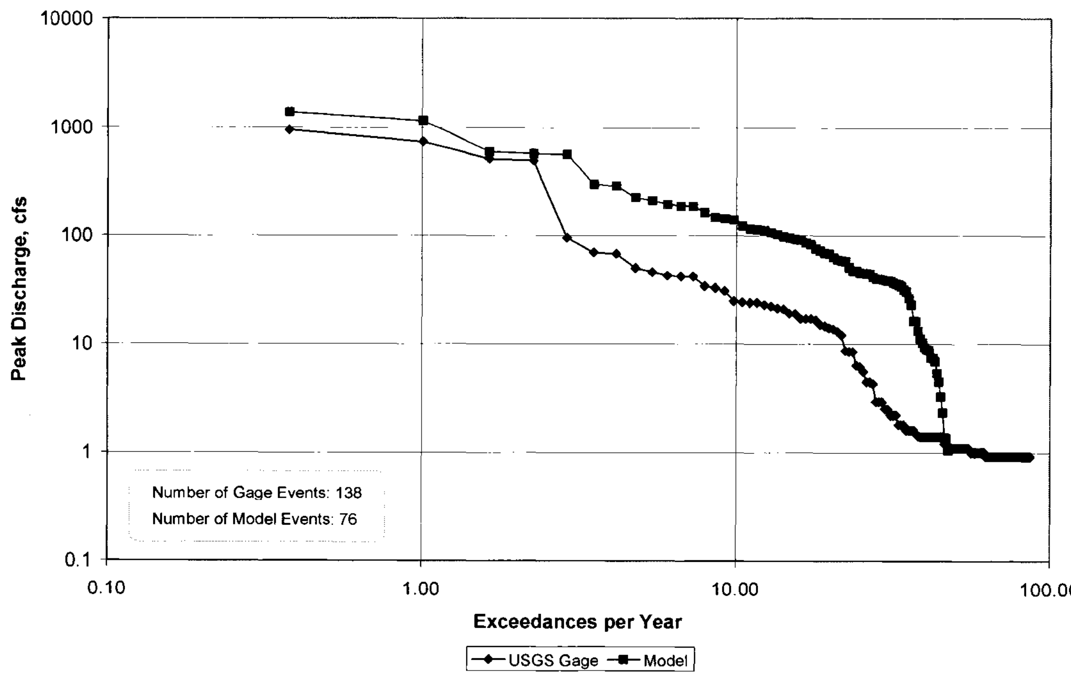


Figure A-8. Peak Flow Frequency Exceedance Curve For Calibration Period: Black Creek.

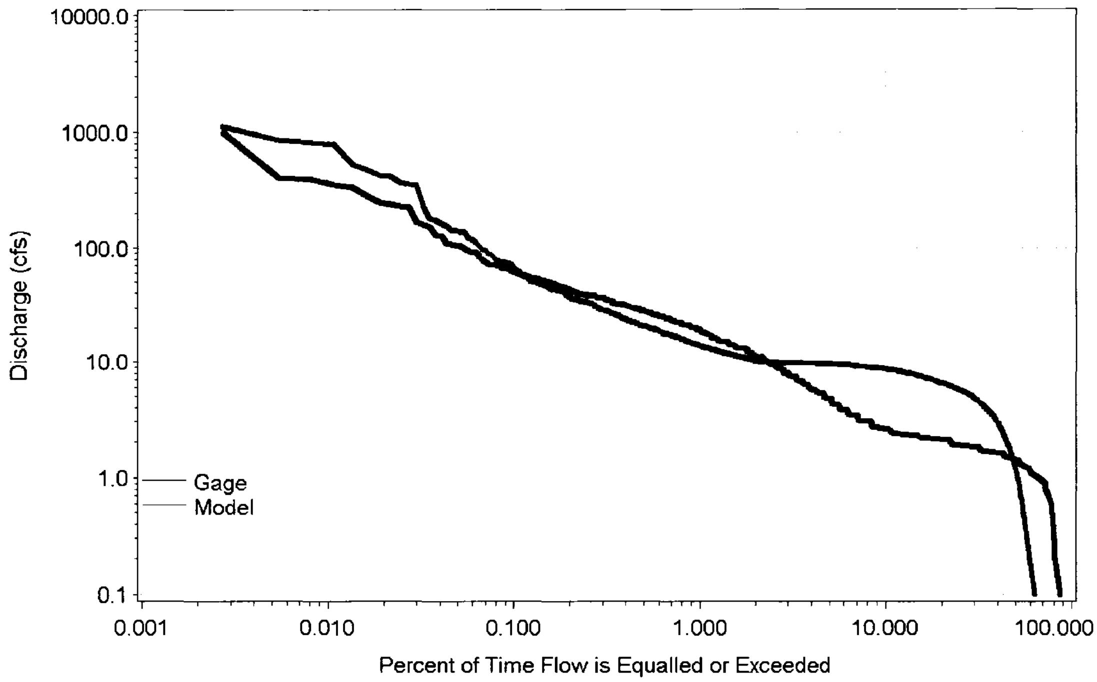


Figure A-9. Flow Duration Curve For Calibration Period: Beaverdam Creek.

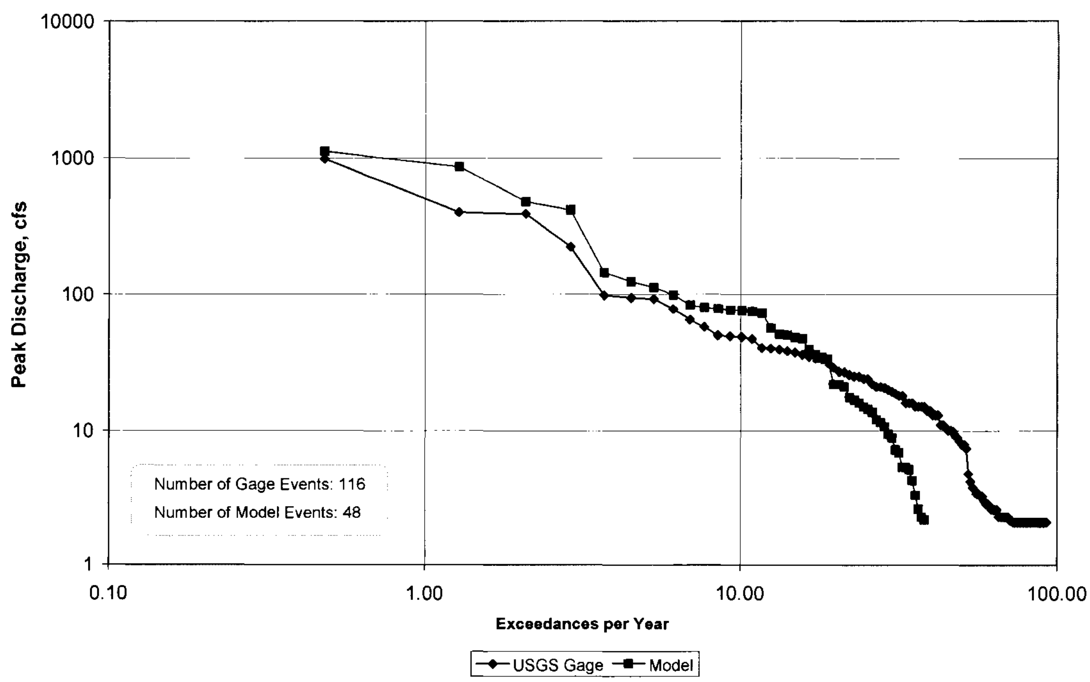


Figure A-10. Peak Flow Frequency Exceedance Curve For Calibration Period: Beaverdam Creek.

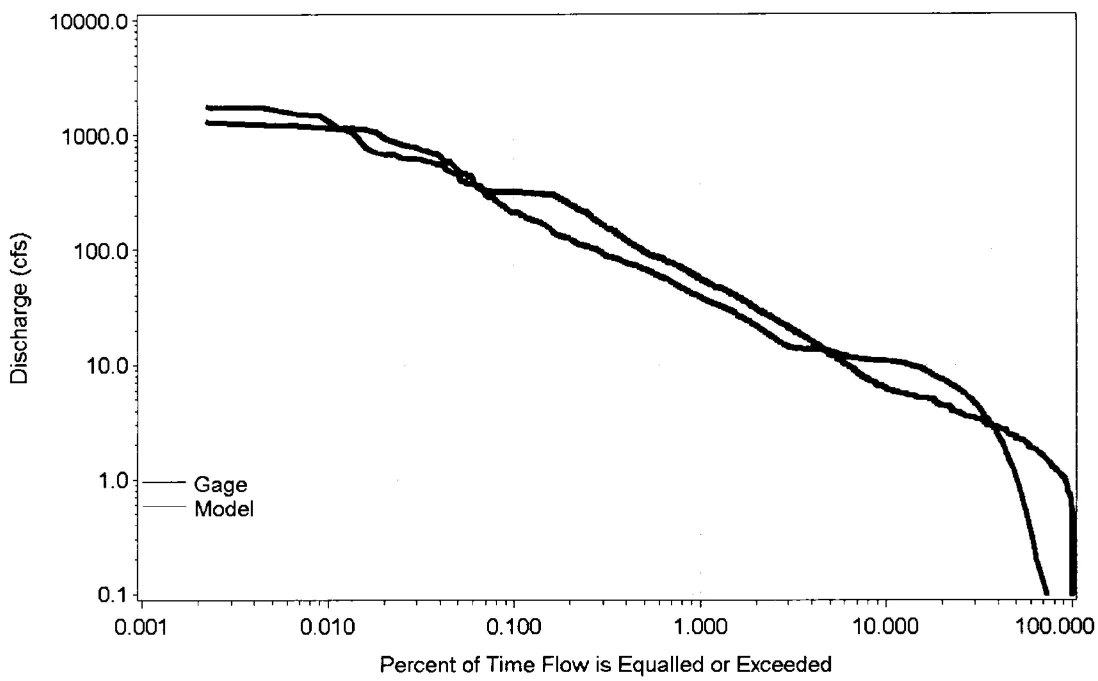


Figure A-11. Flow Duration Curve For Calibration Period: Pigeon House Creek.

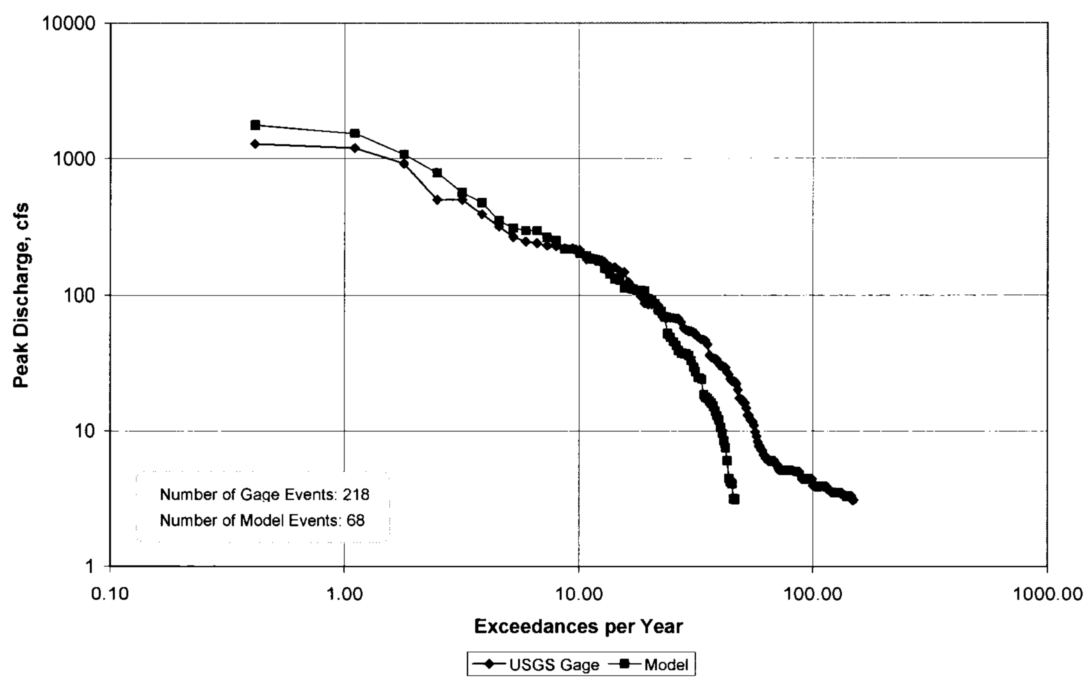


Figure A-12. Peak Flow Frequency Exceedance Curve For Calibration Period: Pigeon House Creek.

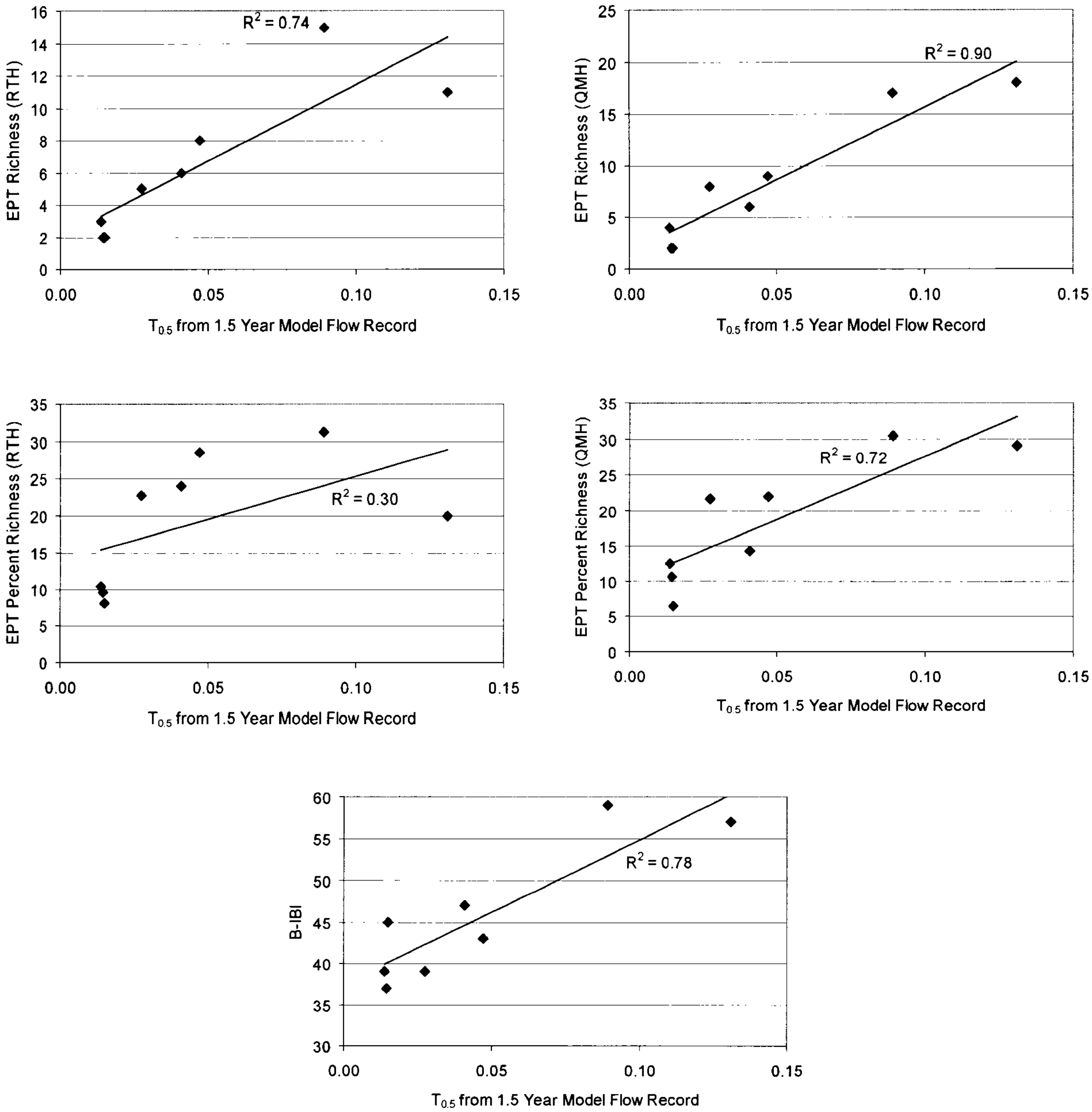


Figure A-13. Relationship of $T_{0.5}$ Calculated From the Calibration Period Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

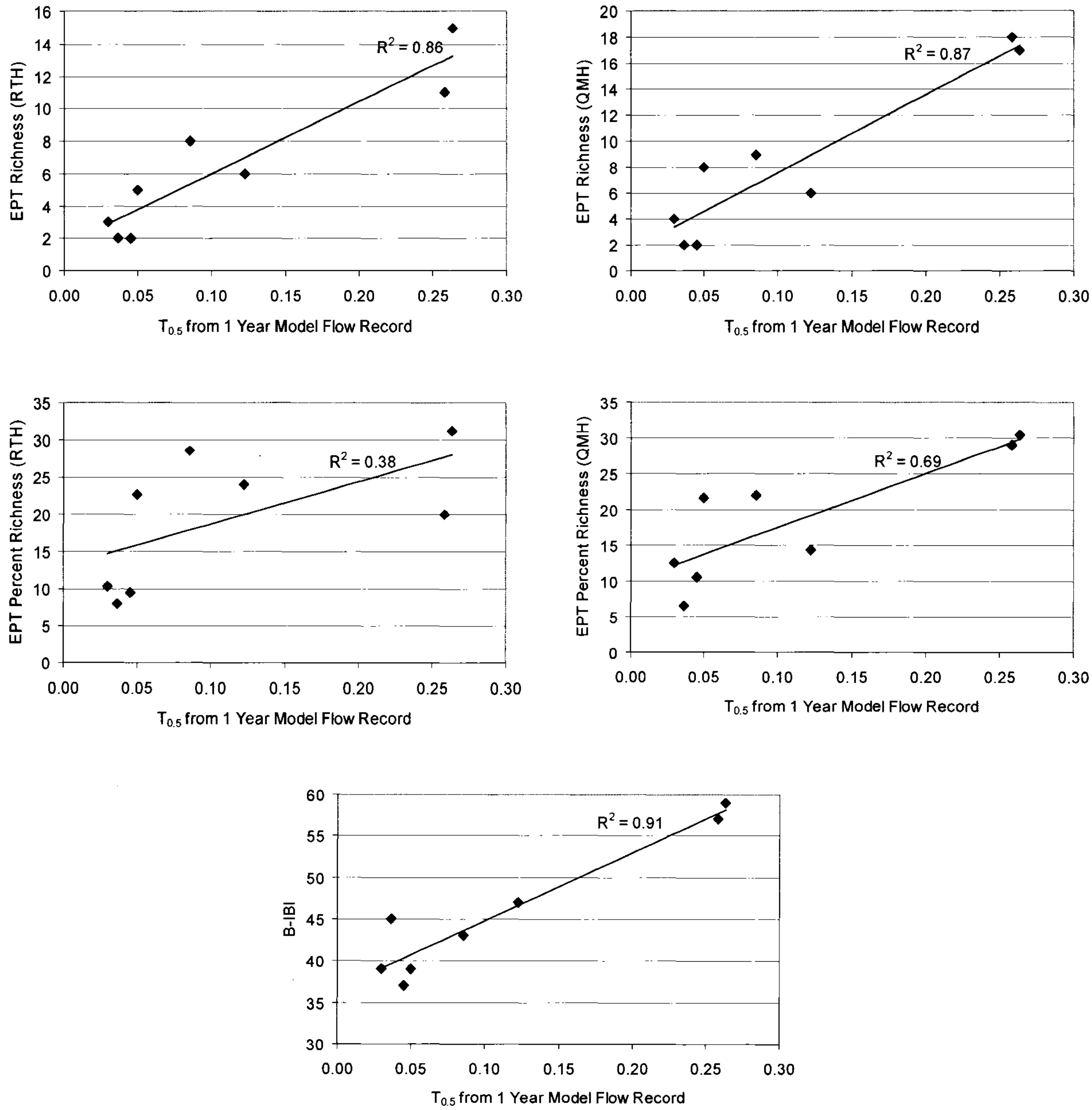


Figure A-14. Relationship of $T_{0.5}$ Calculated From the 1-Year Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

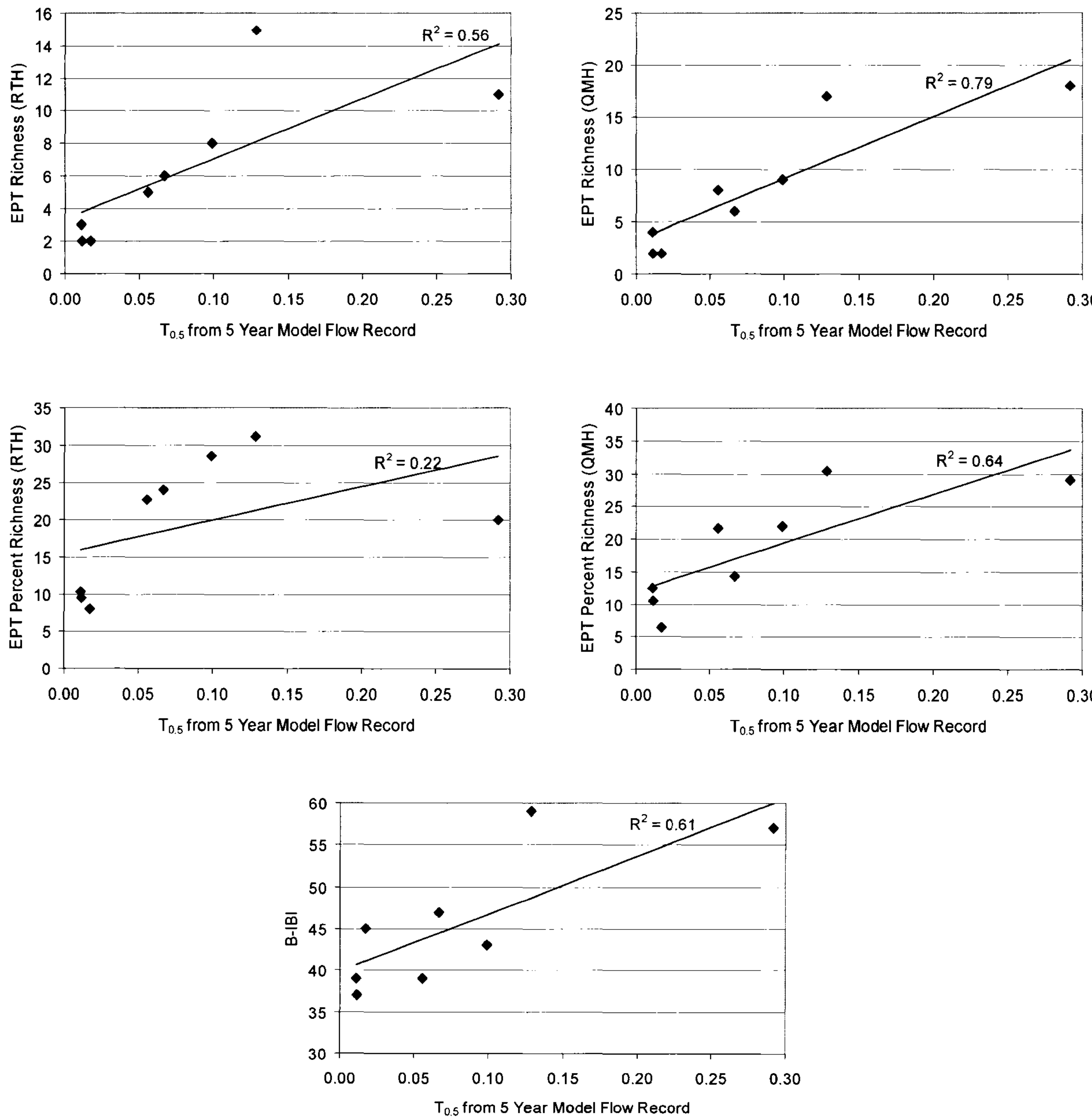


Figure A-15. Relationship of $T_{0.5}$ Calculated From the 5-Year Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

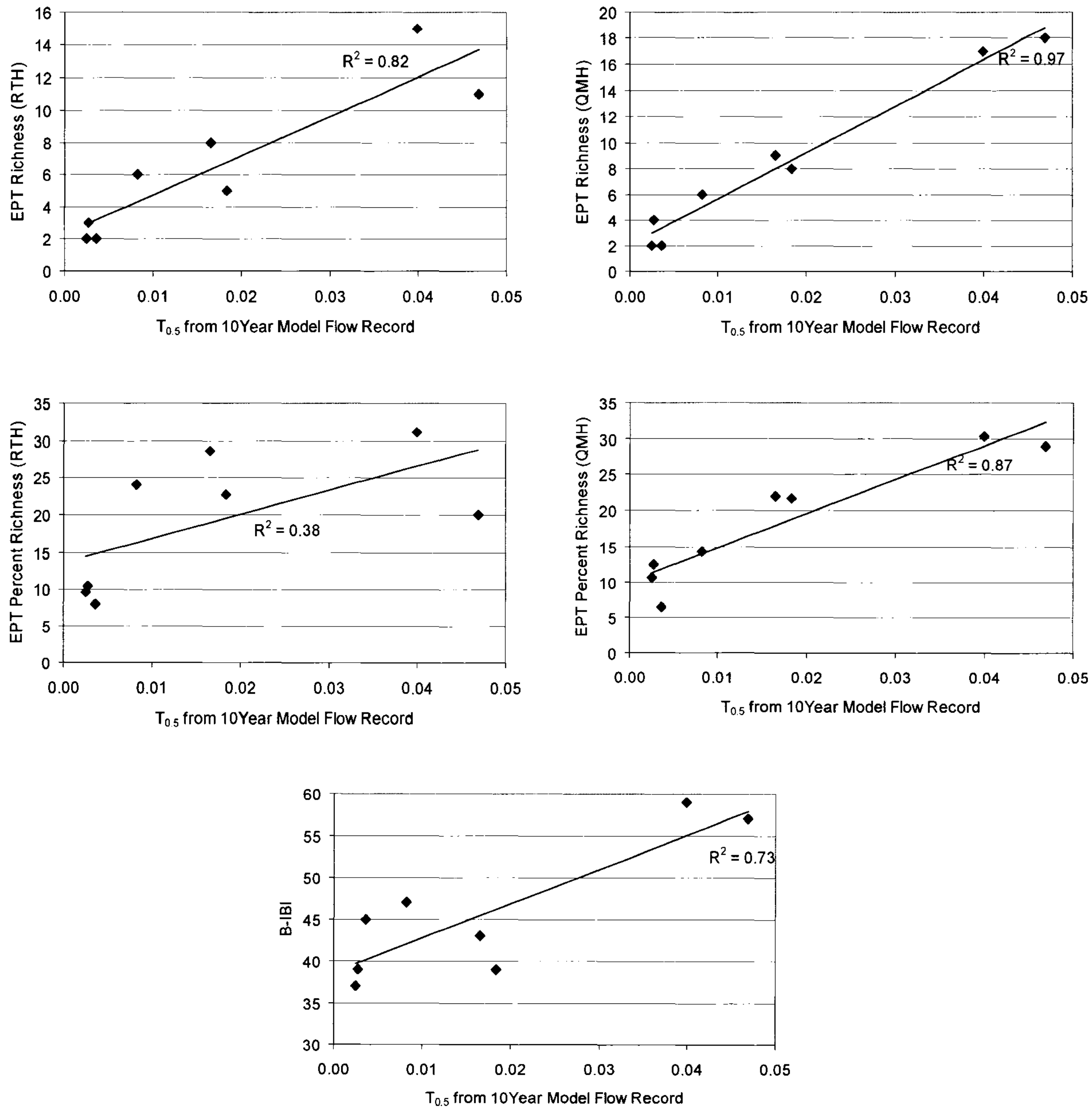


Figure A-16. Relationship of $T_{0.5}$ Calculated From the 10-Year Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

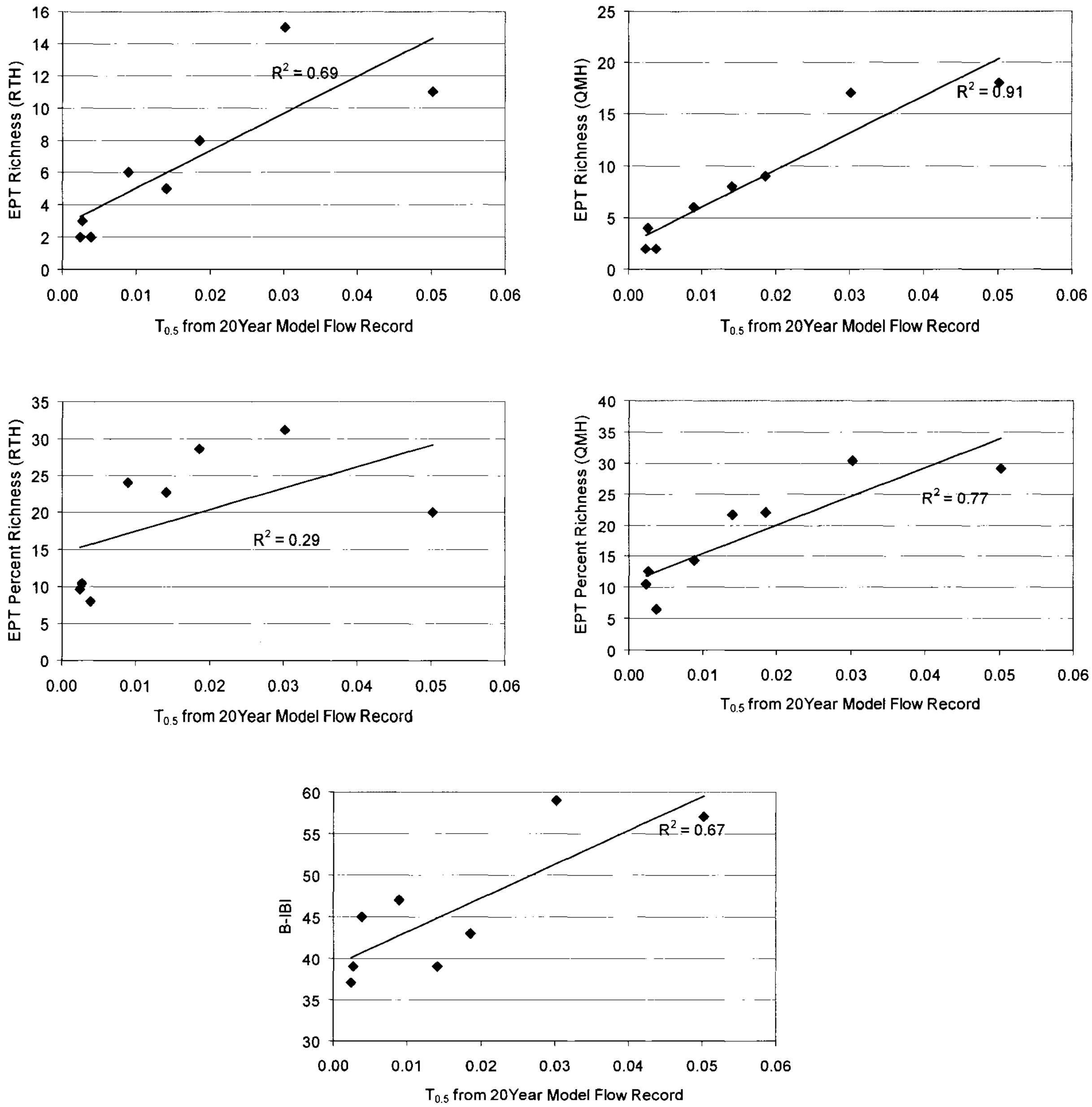


Figure A-17. Relationship of $T_{0.5}$ Calculated From the 20-Year Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

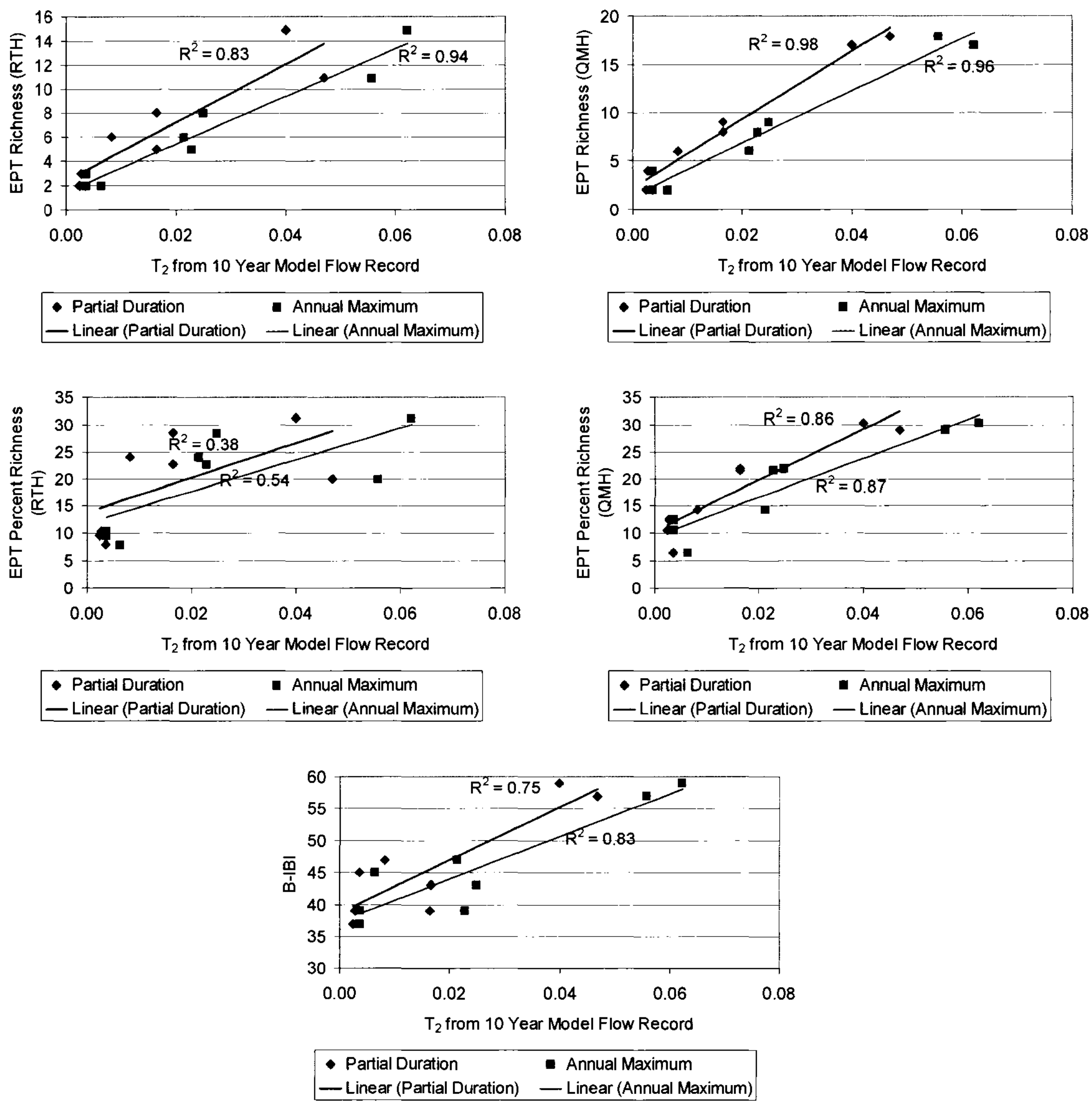


Figure A-18. Relationship of T_2 Calculated From the 10-Year Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).

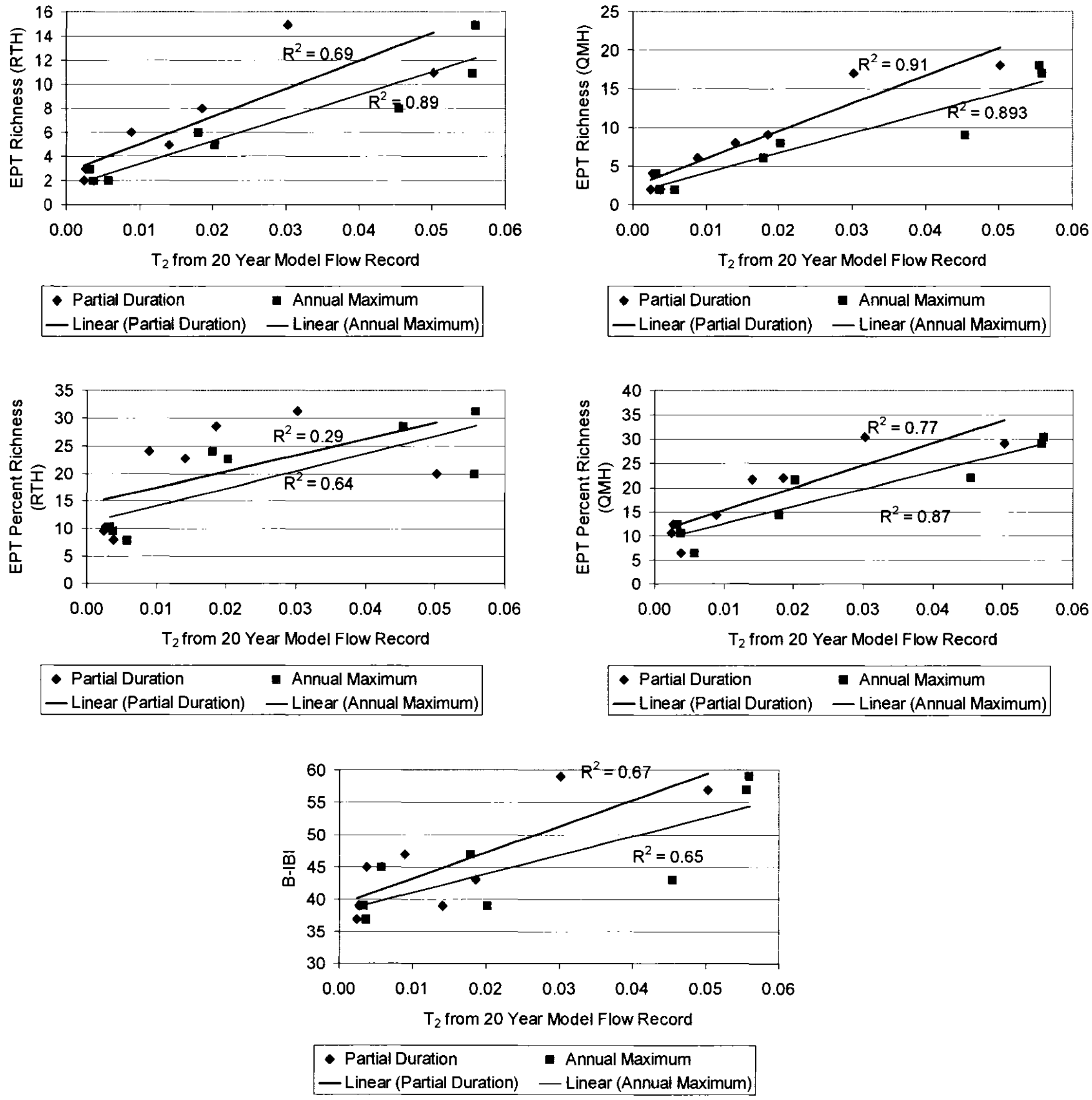


Figure A-19. Relationship of T_2 Calculated From the 20-Year Flow Record to Benthic Macroinvertebrate Sampling Data (a) EPT Richness (RTH), (b) EPT Richness (QMH), (c) EPT Percent Richness (RTH), (d) EPT Percent Richness (QMH), (e) Benthic Index of Biotic Integrity (B-IBI).