LINKING URBANIZATION TO STREAM GEOMORPHOLOGY AND BIOTIC INTEGRITY IN THE LAKE CHAMPLAIN BASIN, VERMONT

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ABSTRACT

The Impervious Cover Model (ICM) was developed to explain the general response of biotic and abiotic characteristics of stream ecosystems to urban impacts over a range of physiographic regions. Much research has shown that a stream ecosystem response can be detected when the total impervious area (TIA) is at or above 10% of the watershed area, however limited research has focused on the impacts of urbanization at different spatial and temporal scales. This study explores these impacts by testing: (1) the effect of TIA on geomorphic stability, physical habitat conditions, and biotic communities at three different spatial scales, (2) the differences between urban and rural downstream hydraulic geometry (DHG) regressions, and (3) the response of biotic communities to different stages of urban channel evolution.

The physical and biotic conditions of stream reaches from 16 small watersheds in northwestern Vermont were assessed and analyzed for a response to TIA at multiple spatial scales. Separate analyses were performed for high and low-gradient stream types, and reach selection criteria minimized the influence of human impacts to channel boundary conditions (e.g., bank armoring) to ensure a robust test of the ICM for upslope TIA alone. Response of geomorphic stability and sensitive macroinvertebrates to TIA was nonlinear and significant ($P < 0.001$), decreasing rapidly at 5% TIA. The effect of TIA on stream condition interacts significantly with drainage area and channel slope ($P < 0.05$). DHG regressions developed for urban and rural watersheds show a significant scale-dependent response ($p = 0.001$) of channel width to urbanization. Analysis of macroinvertebrate data from reaches in different stages of channel evolution indicates that stable reaches support greater EPT richness ($p < 0.005$) and overall species richness ($p < 0.01$) than unstable reaches. Results of ICM analyses and DHG regressions demonstrate that streams in Vermont may be more sensitive than those in other regions of the country in their response to urbanization, and that the response is scale-dependent. Results also indicate that some recovery of biotic communities may be possible following natural channel restabilization.

A separate analysis was performed using urban and rural DHG regression parameters published for different physiographic regions of the United States. DHG parameters α and β describe the intercept and regression slope respectively, and non-parametric tests showed significant differences between urban and rural watershed types using both parameters from channel width equations ($P < 0.005$). These results indicate a consistent response of channel width to urbanization that is dependent on the scale of the watershed area up to areas of ~50 km^2, a scale that includes first- to third-order headwater streams that are often directly impacted by urban and suburban development. The results have important implications for land use planners in urbanizing watersheds and stream restoration professionals intending to use DHG regressions for channel restoration designs. Failure to recognize scale-dependent differences in the response of channel geometry to urbanization could lead to improper channel restoration designs and project failure.

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CHAPTER 1

LITERATURE REVIEW

Introduction

The impact of urban stormwater runoff on small streams in urbanizing environments has become a critical area of research in the field of watershed management (Paul and Meyer, 2001). Many stream ecologists across the country have shown that urban land use has detrimental effects on aquatic biota (Coles et al., 2004; Fitzpatrick et al., 2004; Miltner et al., 2004), and watershed scientists have long known that urban land use increases sediment and nutrient loading to receiving waters (CWP, 2003). Although the deleterious effects of urbanization on the physical structure and habitat conditions of streams are often readily observable, quantifying these effects can be difficult and costly (Pizzuto et al., 2000; CWP, 2003). Nevertheless, a growing body of scientific literature is beginning to show the effects of stormwater runoff on relative sediment yield due to channel instability (Trimble, 1997a; Nelson and Booth, 2002), and on physical habitat conditions (Coles et al., 2004; McBride and Booth, 2005).

This review summarizes recent as well as early published literature on the effect of urban land use on stream ecosystems. Particular attention is paid to the following topics: 1) the effect of urban land use on stream geomorphic stability and physical habitat conditions; 2) the comparison of hydraulic geometry between urban and rural watersheds; 3) the effect of urban land use and geomorphic stability on aquatic biota. At the end of this literature review there is a summary of the scope and goal of this study in the context of the overall body of literature, as well as the Vermont Agency of Natural Resources' (ANR) efforts to mitigate the effects of stormwater runoff in the Lake Champlain Basin through the implementation of Total Maximum Daily Load (TMDL) analyses.

Effects of Urbanization on Stream Geomorphology

Early Studies of Urban Geomorphology

Wolman (1967) was among the first geomorphologists to measure the physical impacts of urbanization on watersheds and stream channels near Baltimore, Maryland. His studies found that average sediment production rates are moderate to high during pre-urban agricultural uses of land, followed by a spike during construction, and finally a decrease in sediment yield after urbanization (Figure 1). Wolman's results also indicate that the channel response following urbanization is a period of deepening followed by lateral migration and channel widening. These observations of the changes in channel form are consistent with other work carried out during the same time period when fluvial geomorphology in the urban environment was a developing science (Hammer, 1972; Nanson and Young; 1981; Graf, 1985). Many of the physical changes observed in these studies are consistent with the expected changes due to urbanization when related to Lane's balance (Lane, 1955), a model that depicts the balance between a watershed's hydrologic and sediment regimes in maintaining a dynamic equilibrium in the channel network (Figure 2).

Graf (1985) observed the effects of rapid urbanization on the fluvial geomorphology of two small watersheds near Denver, Colorado. Graf (1985) found that the initial impact for these sites was extreme aggradation and increased floodplain access due to increased rates of upslope erosion. The secondary impact, after the watershed development was nearly complete and impervious cover in place, was vertical incision and downcutting through the previously aggraded material. Arnold et al. (1985) conducted a similar geomorphic and hydrologic study of a small, urbanizing watershed in Connecticut. The frequency of bankfull discharge increased, and changes in the sediment regime were consistent with Wolman's (1967) observations describing the effects of urbanization.

In contrast, Hammer (1972) found that stream channels in watersheds near Pennsylvania tended to first enlarge when responding to urbanization. Hammer's enlargement ratio refers to the relative increase in channel size due to urbanization where rural stream channel dimensions are used as the basis for this change. Of all the variables Hammer (1972) tested, basin slope and level of urbanization were found to have the greatest effect on enlargement ratios of cross sectional areas. The age of urbanization was also found to significantly predict enlargement ratios, where newly urbanized areas (less than 4 years) were found to have little enlargement when compared with older urbanized areas (greater than 4 years).

The effect of urbanization on the watershed-scale geomorphic processes may differ across physiographic regions, as indicated by these studies summarizing results from the Rocky Mountains, the Piedmont, New England. While a number of the early studies of geomorphic response suggest that aggradation is the primary process set in motion by urbanization (Wolman, 1967; Graf, 1985), other more recent studies suggest that degradation and incision processes dominate (Booth, 1990; Pizzuto et al., 2000), as observed by Hammer (1972) decades ago. These studies also indicate that the time period, legacy effect of previous land uses and scale of urbanization determine whether the initial channel response will be degradation (incision) or aggradation. In urbanizing watersheds that were previously dominated by agricultural land uses, degradation will become the dominant process as the stream channel network vertically adjusts through the aggraded material. In addition, rapid urbanization which occupies a majority of the watershed may, in a matter of a decade, overwhelm any effects from past land uses (e.g., agriculture) and cause severe degradation and channel enlargement. It is also important to point out that Wolman's model may better represent how watersheds in this time period (1960's & 1970's) responded to rapid urbanization in the absence of current-day policies that vigorously control the extreme sedimentation associated with construction.

Recent Literature Reviews of Urban Geomorphology

A number of excellent reviews published within the past five years have compiled previous work on this topic. The Center for Watershed Protection (CWP, 2003) provided a review of the negative effects of impervious cover on the physical integrity of streams, including: physical habitat; channel enlargement and erosion; embeddedness of substrate; temperature; and large woody debris. The CWP (2003) notes the difficulty in defining and measuring "habitat integrity" as an obstacle for further study. Habitat variables typically consist of a mix of quantitative and qualitative measurements for a suite of physical components at the reach scale, and when these components are given equal weight (rather than developed from a statistical analysis or a predictive model) they can fail to discriminate watershed effects. This is an important point given that ANR's Stream Geomorphic Assessment (SGA) Protocols (VTDEC, 2005) incorporate composite metrics for evaluating geomorphic stability (RGA) and physical habitat conditions (RHA).

Paul and Meyer (2001) reviewed the physical response of stream channels to urbanization and reported a similar trend to Wolman's (1967) model of aggradation followed by degradation and channel enlargement. Chin (2006) reviewed over 100 studies spanning 50 years, all of which address the impact of urbanization on stream ecosystems. This review highlights the importance of understanding spatial and temporal variability of the aggradation and degradation phases in the context of management strategies. Urban watershed management and stream channel restoration efforts which fail to recognize predictable patterns of change brought on by urbanization will be more likely to fail.

Brabec et al. (2002), whose review approaches the problem from a land use planning perspective, summarized the methodologies most commonly used to calculate effective impervious area (EIA) and total impervious area (TIA). TIA is the total impervious area found in the watershed regardless of the size, location, and connectivity of the patches which contribute to the total. EIA is the fraction of TIA which is hydrologically connected to the watershed drainage network via man-made infrastructure such as catch basins and culverts. This study revealed that there are many inconsistencies in how watersheds scientists measure EIA and TIA and that calculations based on zoning units (e.g., parcel size) vary significantly across geographic areas. To make the intersection of watershed science and land use policy more productive, Brabec et al. (2002) promote the development of standard regional methodologies for calculating TIA and EIA which can be verified at the local scale using GIS-based mapping.

Regional Responses to Urban Impacts

Much of the recent literature published on the effect of urbanization on fluvial geomorphology has come out of the Pacific Northwest region. This is partly attributable to the fact that salmon species (*Oncorhynchus spp.*) found in this region are threatened or endangered by land use changes (Polos et al., 1999) and the protection of critical habitat for their survival involves extensive assessment of geomorphic condition. Although the underlying geologic conditions in this region are different from those found in Vermont, many of the geomorphic responses observed in the urban setting are applicable to our region.

Booth's (1990) initial approach to measuring and predicting stream channel incision in small streams outside of Seattle is an example of how methods from this region could be applied to Vermont. He found that channel slope and underlying geology are two critical characteristics that explain where catastrophic incision occurs. Booth and Jackson (1997) followed up on this study with a modeling analysis in the same geographic area that sought to understand the effects of impervious cover on alterations to the hydrologic regime. This study used a continuous simulation model called Hydrologic Simulation Program-FORTRAN (HSPF) to demonstrate that detention ponds designed to local standards are ineffective at mitigating watershed development. Booth and Jackson (1997) discuss EIA watershed threshold of 10%, and suggest this level may be an inadequate measure of impact since aquatic life degradation can be detected at levels below 10%.

While the measurement of impervious cover is clearly a key component in urban watershed assessment (Paul and Meyer, 2001; Brabec et al., 2002; CWP, 2003), some more recently published studies from the Pacific Northwest have shed additional light on the relative importance of impervious cover as a “stand-alone” surrogate for watershed health (Booth et al., 2002; Nelson and Booth, 2002). Building on the hypothesis that the altered hydrologic regime is the key driver of physical impairment, Booth et al. (2002) propose that in rapidly urbanizing watersheds other measures, such as the protection of forested lands in headwaters watersheds, may be more effective at avoiding stream channel erosion than reliance on structural BMPs for mitigation. This approach is relevant to assessing Vermont’s stormwater-impaired streams because other variables, not simply impervious cover, may also explain why some channels exhibit severe adjustment in watershed zones with little urban land cover. Legacy factors to previous land use and channel alterations that need to be kept in mind have been pointed out in studies from the Pacific Northwest (Nelson and Booth, 2002). In Vermont, these additional variables might include: 1) previous agricultural impacts, 2) channel straightening and encroachment in the urban environment, and 3) beaver ponding.

The studies from the Pacific Northwest point towards some key conclusions that are relevant to the assessment of stream geomorphic stability and management of stormwater runoff in urbanizing watersheds in Vermont, including: 1) watershed scientists should be cautious in the use of impervious cover alone as a surrogate of stream/river health (Booth et al., 2002; Booth et al.; 2004); 2) hydrologic metrics that reflect chronically altered streamflows may provide a better, and more direct link between watershed changes due to urbanization and degrading biologic condition (Booth et al., 2004); 3) there is little evidence to suggest that the mitigation of non-hydrologic factors (e.g., restoration of channel geometry) can remediate the consequences of urban development (Booth, 2005).

Outside of the Pacific Northwest, there are a limited number of studies that describe the geomorphic impacts of urbanization with the same detail as the research summarized above. Although a small body of literature is building in this field in the Northeastern U.S. (Pizzuto et al., 2000; Coles et al., 2004; Cianfrani, 2005), there is a significant gap of scientific knowledge in this region. The difference in background geologic conditions, as well as historic land use patterns, suggests that the response of the fluvial system to urban impacts in our region may be different from that observed in the Pacific Northwest.

Coles et al. (2004) present an extensive analysis of the effects of urbanization on 29 medium-sized watersheds in eastern Massachusetts, New Hampshire, and Maine; representing the most thorough study of its type in New England. The results of this study agree with other nationwide observations that the integrity of aquatic biota and water chemistry decline as urbanization increases (Stepenuck et al., 2002; Sheeder and Evans, 2004). Results of geomorphic change in this study show that bankfull depth increases and that bankfull width to depth ratio decreases as urbanization increases. Coles et al. (2004) summarize that the streams are systematically narrower and deeper with increasing urban intensity, however give no explanation for why this might be. It could be that in highly urbanized areas of this study, streams are increasingly channelized and therefore their narrow channel form is human-induced by direct encroachment, rather than upslope watershed processes.

The results from Coles et al. (2004) contrast with another similar study carried out in the Piedmont region of Southeastern Pennsylvania (Pizzuto et al., 2000). Pizzuto et al. (2000) analyzed eight paired urban and rural watersheds to assess the impacts of urbanization on fluvial morphology of small streams. Although most of the results agree with what has been observed in the Pacific Northwest (Booth, 1990; Booth et al., 2002), there have been very few quantitative studies of paired urban and rural streams like this study. One interesting result of this study is that distributions of bed sediment grain sizes did not differ significantly between the two stream types, suggesting that (despite the greater stream power in urban systems) there is an upland source of coarse-grained sediment in the urban systems, possibly from extensive bank erosion. Pizzuto et al. (2000) suggest that there continues to be a significant input of coarse and fine sediment to the urban stream channels via bank erosion and mass failures. These inputs are maintaining sediment production and transport rates similar to those observed in the rural channels. Cianfrani (2005) built on the Pizzuto et al. (2000) dataset by comparing watershed TIA with channel dimensions for the same watersheds. Cianfrani (2005) found that the TIA alone was a poor predictor of channel geometry, while a multiple regression including other reach-scale variables (vegetation type, bankfull width to depth ratio, LWD) was about twice as powerful at predicting channel dimensions.

Doyle et al. (2000) conducted a more qualitative study on seven urban stream channel sites in Central Indiana. Unlike Pizzuto et al. (2000), Doyle et al. (2000) report that urbanization was not distinguishable as a driver of overall instability (as measured by a composite metric similar to RGA), yet certain instability metrics were associated with levels of urbanization. These metrics included higher values for channel incision, channel widening, and lower values for large woody debris (LWD). Lower quantities of LWD have also been reported in other studies of urban watersheds (Finkenbine et al., 2000; CWP, 2003).

Sediment Budgets in Urban Environments

Sediment budgets in the urban landscape also represent an attempt to understand the effect of land use change on the dynamic equilibrium of small streams. Often a key water resource management concern in urbanizing watersheds is the increased sediment loading and discharge caused by changes to land cover. Some studies have shown that the majority of sediment yield from urbanizing watersheds is from streambank erosion (Trimble, 1997a), while others show that legacy effects of past land uses still override the negative effects of sprawling urban land (Nelson and Booth, 2002). This uncertainty has recently come into play in important stormwater management policy decisions involving urbanizing watersheds in Vermont (VWRB, 2004), and presents a challenge in many states across the country.

As previously noted, Wolman (1967) was one of the first geomorphologists to address the topic of sediment budgets in urban environments. Wolman's pioneering observations of changes to channel form have been duplicated by other researchers in similar settings over the years (Doll et al., 2002; Neller, 1989; Pizzuto et al, 2000). Neller (1989) compared channel cross-sectional areas in urban and rural watersheds in Australia and found that urban channels were on average about four times larger than rural channels, equating to an enlargement ratio of 4.0. This widening and deepening of the channel during urbanization leads to severe bank erosion resulting in sediment and nutrient inputs to the watershed. Doll et al. (2002) and Pizzuto et al. (2000) also calculated enlargement ratios to determine the effect of urbanization on channel geometry, where this ratio was found to be 2.65 and 1.8 respectively.

Quantitative observations of bank erosion and increases in channel capacity are the basis for two sediment budgets carried out in urbanizing watersheds to determine the relative contribution of endogenous and exogenous sources of sediment (Nelson and Booth, 2002; Trimble, 1997a). In a 10 year sediment budget, Trimble (1997a) found that two-thirds of the sediment yield in the San Diego Creek comes from streambank erosion, and one-third from surface runoff. On the other hand, Nelson and Booth (2002) found that the effects of gravel roads associated with remnant forestry practices still accounted for the vast majority of sediment delivery in a rapidly urbanizing watershed.

Despite the thorough "accounting" done for each of these budgets, there still remain important questions about the fate of sediments produced either from bank erosion or runoff. These questions are relevant to stormwater management in Vermont, and include: 1) What temporal scale is adequate for the measurement of urban-induced sediment discharge (either suspended or bedload) from a watershed, and how does legacy sediment from past deposition confound these measurements? 2) What role does resuspension of fine sediment play in total discharge measurements? 3) What are appropriate background levels of bank erosion and overall sediment discharge from "reference" watershed to use as benchmarks for comparison?

Spatial and Temporal Scale of Urban Impacts

With the exception of one urban watershed (Trimble, 1997a), much of the research in the urban environment is dominated by studies that look at a "snapshot" of physical conditions at the time of the survey. In addition to limitations with temporal scales, recent literature has highlighted the shortcomings of approaching land use and stream geomorphology interactions at one spatial scale (Chin and Gregory, 2005; McBride and Booth, 2005). Chin and Gregory (2005) outline a data collection and management approach to urban-induced geomorphic change, whereby geomorphic stability measurements are collected throughout multiple adjacent subwatersheds and the degree of channel adjustment is mapped at the reach scale, providing a framework for determining the cause of channel adjustment in urbanized areas.

McBride and Booth (2005) also assessed the impacts of land use at different spatial scales in small, urbanizing watersheds in the Puget Sound. They found that the best explanation of geomorphic integrity was a combination of the percentage of intense and grassy urban land in the subwatershed and the percentage of intense and grassy urban land within 500 m of the site ($R^2 = 0.52$, $p < 0.0005$). These results, which show that local scale variables were important predictors of channel condition, indicate the importance of considering local land cover in addition to the complete upstream drainage area land cover. Recent research has also determined that stream crossings (bridges and culverts) have a significant negative effect on geomorphic condition (Chin and Gregory, 2005; McBride and Booth, 2005), indicating that the location and scale of channel instability can be directly tied to direct impacts to channel boundary conditions.

Hammer (1972) first considered the importance of the temporal scales associated with urbanization and its effects on stream condition. Finkenbine et al. (2000) emphasize the importance of determining the channel evolution stage (Schumm, 1977) of urban streams before restoration strategies can be applied. Finkenbine et al. (2000) estimate that urban streams in the Vancouver, Canada area may reach new equilibrium conditions within 20 years. Henshaw and Booth (2000) found that streams in the nearby Puget Sound region also stabilized within one to two decades in the absence of additional urbanization. McRae and D'Andrea (1999) developed a model to predict the timeframe of channel enlargement and found that 50 to 60 years may pass before quasi-equilibrium (Schumm, 1977) channel dimensions are achieved in the Southern Ontario region. ANR's SGA protocols (VTDEC, 2005) include methods for determining channel evolution stage, which is important in understanding the temporal scale of adjustments in my study watersheds. Nearly all of the measurements included in the SGA protocols represent a "snapshot" in time; however channel evolution stage observations provide important insight into the physical processes that that shape channel form and habitat while a stream is under adjustment. Measuring the response of biotic communities to channel evolution stages is an example of how temporal scales can be incorporated into my research, and this is explained in more detail in Chapter 2.

Urban Downstream Hydraulic Geometry Regressions

Downstream hydraulic geometry (DHG) regressions were first developed by Leopold and Maddock (1953) as a method to quantify the differences in channel geometry and velocity over a range of bankfull discharges and watershed areas. The equations used to develop DHG regressions are simple power law functions, and an example for channel width is provided in equation 1:

$$W = \alpha Q^{\beta} \qquad (1)$$

where *W* is the bankfull channel width (m), *Q* is bankfull discharge (m^3/s), α is the regression coefficient, and β the regression exponent (and the slope of regression line). Leopold and Maddock's (1953) original work established the foundation for the current use of DHG regressions in river restoration projects (Rosgen, 1994; Rosgen, 1996). However, many researchers have reported on the limitations of DHG regressions (Wohl, 2004; Stewardson, 2005), and their value for stream restoration designs has been seriously questioned by many geomorphologists (Kondolf et al., 2001; Wohl et al., 2005).

In an effort to better understand how bank vegetation influences channel form, several researchers have shown significant differences between forested and non-forested channels, whereby channel width and cross-sectional area is greater in forested channels than non-forested channels (Trimble, 1997b; Davies-Colley, 1997; Hession et al., 2003; Anderson et al., 2004). This information is useful for describing the underpinning water-soil-vegetation processes and in the consideration of bank vegetation in channel restoration designs. However, similar process-based explanations of urban-rural channel geometry differences are scant in the literature.

The CWP review (2003) includes narrative about what is assumed to be the typical long-term channel response to urbanization: stream channels enlarge in cross-sectional area to accommodate more frequent channel-forming flow events. This effect is well understood in the context of Bagnold's (1966) original physical analysis of stream power and its ability to transport bed substrate. Although a number of studies previously described in this review do quantify this effect (Wolman, 1967; Hammer, 1972; Graf, 1985; Pizzuto et al., 2000; Coles et al., 2004), only a handful of studies use DHG regressions to mathematically quantify this effect across a range of drainage areas. Within these studies, there is a consistent difference between urban and rural channel geometry, especially in channel cross-sectional area and width (Allen and Narramore, 1985; Neller, 1989; Doll et al., 2002; Hession et al., 2003).

Neller (1989) showed that urban channel cross-sectional areas are systematically larger than their rural counterparts and that this effect is magnified at smaller drainage areas in Australia. Allen and Narramore (1985) showed this same effect with channel width in two different lithologies of shale and chalk-bottomed streams. Hession et al. (2003) studied 16 rural and 10 urban streams in the Piedmont region of Pennsylvania, Maryland and Delaware, detailing the differences in hydraulic geometry between the two stream types and their bank vegetation types. They also show that the slopes and intercepts of the DHG regressions equations were significantly different between the two land use settings. Doll et al. (2002) also found that urban streams are significantly wider and larger in cross-sectional area by comparing channel geometry in a study of 17 urban and 13 rural streams in the Piedmont region of North Carolina.

As a whole, these studies comparing DHG regressions in urban and rural watersheds show at least two consistent patterns: 1) the slopes of urban regressions for width and cross-sectional area are less than rural slopes (suggesting a scale-dependent response of channel geometry to urbanization); 2) there is greater variance in the urban regressions, suggesting better predictability of rural models (the coefficient of variance, or r^2, is higher in rural curves). These patterns are the focus of a trans-regional analysis of DHG regressions in Chapter 3.

Effects of Urbanization on Aquatic Biota

Response of Biota to Urban Stressors

Numerous studies in recent years have shown the negative effect of urbanization on aquatic biota, and a recent review published by the CWP (2003) found over 33 studies showing this effect on aquatic insect communities (macroinvertebrates) and 19 on fish communities. Much of this research has focused on the impact of urbanization on aquatic communities that are sensitive to human impacts, using indices such as EPT richness (Stepenuck et al., 2002; Roy et al., 2003). Other studies have used overall macroinvertebrate indices (Morely and Karr, 2002; Fitzpatrick et al., 2004; Miltner et al., 2004; Long and Schorr, 2005). Although some studies have shown the impact of urbanization on certain habitat features important to aquatic communities such as LWD (Horner et al., 1996; Finkenbine et al., 2000) and the geometry of geomorphic features like pools and runs (Pizzuto et al., 2000), little research exists which shows the influence of geomorphic stability on aquatic communities (Sullivan et al., 2004).

Stepenuck et al. (2002) looked at the effects of TIA on a number of biotic indices in Wisconsin streams. They found a negative correlation for all biotic indices with impervious cover, with threshold levels of impervious cover corresponding to significant biotic declines between 8% and 12%. Miltner et al. (2004) used fish assemblages in Ohio watersheds from 267 sampling locations and found that biotic integrity declined significantly when impervious cover exceeded 13.8%, and fell below reference biotic conditions when impervious cover exceeded 27.1%. Both Stepenuck et al. (2002) and Miltner et al. (2004) focused on one spatial scale, using complete upslope watershed land use. Similarly, Long and Schorr (2005) studied the correlation between watershed urban land use and biotic integrity as measured by fish assemblages (IBI) in streams found in the Tennessee River Watershed, and found that IBI scores and diversity were negatively correlated with watershed urban land use. Long and Schorr (2005) reported values for correlation coefficients that are within the range (0.5 to 0.7) observed for similar analyses from other studies such as Fitzpatrick et al. (2004) and Morely and Karr (2002).

Other studies compare the effect of TIA on biota across different spatial scales to determine the relative effect of upslope watershed land use versus local, reach-scale land use (within buffer or corridor). Morely and Karr's (2002) work in the Puget Sound area on 2nd and 3rd order streams compared urban land cover (based on 1998 satellite imagery) with biotic integrity indices (IBI) at various scales. Across all study sites, IBI declined as urban land cover increased at the upslope watershed scale (r = -0.71). Certain reaches, however, showed that local land cover measured within a 200 m buffer was better correlated with biotic indices. This finding supports the idea that local-scale urbanization may be important for subwatersheds and reaches where EIA represents a large fraction of TIA. Potter et al. (2005) also considered multiple scales of urban land use in predicting macroinvertebrate community structure in North Carolina. Their results indicate that the best predictor of biotic integrity to be topographic complexity of the watershed at both scales, showing that those areas with greater complexity are where less development and agriculture is found.

Biotic Response to Geomorphic Adjustments

Aquatic ecologists have long understood that physical habitat characteristics are important drivers of the diversity, abundance, and distribution of riverine biota (Maddock, 1999). A growing body of literature suggests that the maintenance of ecological integrity (Angermeier and Karr, 1994) in riverine ecosystems involves the complex interaction of many physical and biological processes at multiple spatial scales (Thompson et al., 2001; Poole, 2002; Richards et al., 2002; Aarts et al., 2004). While some studies have begun to link geomorphic processes with ecological integrity at the reach scale (Sullivan et al., 2004; Cianfrani, 2005; Sullivan et al., 2006), others have shown that there are many other physical and biotic processes interacting which may make this link difficult to quantify (Thompson et al., 2004).

The physical processes involving a balance between water and sediment inputs (Lane, 1955) that occur in river corridors are inherently dynamic (Schumm, 1977), and these multi-scaled processes create and maintain the heterogeneity in physical habitats that support diverse biotic communities. Despite the knowledge of these processes, few studies have attempted to link geomorphic stability measurements to biotic integrity. Long and Schorr (2005) found that an IBI measure of fish assemblages was negatively correlated with erodible streambank, suggesting a link between geomorphic stability and biotic integrity in Tennessee watersheds. Fend et al. (2005) studied upland and valley streams in the Santa Clara Valley, California, and found that measurements describing physically modified channels that were unstable or previously unstable (e.g., percent of channel armored) were more important than upslope urban land cover in predicting biotic richness.

Studies on this topic in Vermont include Cianfrani's (2005) assessment of the impacts of local and watershed land use conditions on different stream biota (fish, macroinvertebrates, and birds). Cianfrani's (2005) results indicate that macroinvertebrates were found to be most sensitive to a combination of local-scale conditions (e.g., vegetation) and the watershed's sediment regime. Sullivan et al. (2004) carried out a similar study using RGA and RHA measurements from paired stable and unstable reaches. Although Sullivan et al. (2004) reported that stable reaches did not support significantly greater macroinvertebrate densities than unstable reaches, EPT richness was significantly correlated ($p < 0.025$) with geomorphic stability. Sullivan et al. (2006) also reported on the influence of geomorphic condition on fish communities in separate study sites in Vermont. Results from this study indicate that geomorphic stability was a significant predictor of fish communities, and suggest that fish may be more responsive to geomorphic stability than macroinvertebrates due to differences in habitat scales. The results of each of these Vermont studies indicate that there is likely a link between geomorphic stability and biotic integrity that warrants further study.

On the other hand, Clark's (2006) recent research on this topic in Vermont has shown that there is little correlation between ANR's RGA scores and measured biotic diversity and richness. ANR has recognized the shortcomings of the composite RHA metrics and is currently working on a revised methodology based on recently published literature (Kline, 2006).

Study Scope and Goal

Contribution to Field of Study

The CWP's (2003) extensive review of the impacts of impervious cover on aquatic systems found that studies on aquatic biota outnumbered those on geomorphic condition by a 2:1 ratio. This ratio is far exceeded when considering studies on sediment and nutrient loading in urbanizing watersheds. A growing body of literature is beginning to quantify the effects of stormwater runoff on relative sediment yield (Trimble, 1997a), channel instability (Booth, 1990; Nelson and Booth, 2002), and physical habitat conditions (McBride and Booth, 2005). These published studies, however, have been focused primarily in watersheds on the West Coast of the U.S. Although the results from a small number of studies have shown these same effects in watersheds of the East Coast of the U.S. (Pizzuto et al., 2000; Coles et al., 2004), little research has been conducted on this topic in New England and no published studies have specifically focused on this effect in the state of Vermont. My thesis research will contribute to this field of study by addressing the impacts of stormwater runoff on the geomorphic stability of small streams in an understudied physiographic region.

With the exception of the study carried out by Coles et al. (2004), my review of the literature found no models that link urban land use to stream geomorphic and physical habitat condition in the New England region. My research differs from the Coles et al. (2004) study by including a much more extensive analysis of physical parameters at multiple spatial scales. It has been noted that very few models of this type have been explored at the national level (CWP, 2003). Amongst the models that have been developed to predict geomorphic condition based on land use, only a few have considered land use at various spatial scales in the watershed (Chin and Gregory, 2005; McBride and Booth, 2005). The development of a GIS model to explore this link using the SGA data from Chittenden County will be useful not only to fill the gap in this area of research regionally and nationally, but also for ANR's efforts in addressing stormwater runoff in Vermont (VWRB, 2004). Further exploration of this relationship at different spatial scales will also help fill a significant gap in the research.

The amount of resources invested into stream restoration projects has risen exponentially since the mid 1980's (Bernhardt et al., 2005), yet many scientists have cautioned that a large percentage of these are carried out with minimal scientific context (Wohl et al., 2005). DHG regressions are important tools in the science of fluvial geomorphology that have been promoted for use in channel restoration (Rosgen, 1996). While DHG relationships are useful in understanding the changes to channel geometry at different scales and as indicators of a watershed's sediment and hydrologic regimes, researchers have reported their misuse in practice through the improper design of channel dimensions with inadequate knowledge of watershed-scale process dynamics (Kondolf et al., 2001; Wohl et al., 2005). Despite the fact that stormwater management is often a goal of channel restoration in urban watersheds (Bernhardt et al., 2005), relatively few studies have used DHG regressions to compare urban and rural channels in the same physiographic region (Allen and Narramore, 1985; Doll et al., 2002). My research will contribute an additional study of urban and rural DHG regressions to this field.

In addition, despite the long-held assumption that geomorphic state strongly influences stream biotic communities, few studies have made this direct link (Sullivan et al., 2004; Sullivan et al., 2006). This research will contribute to the understanding of the influence of geomorphic state on macroinvertebrate communities in the urban setting.

Application of Research in Vermont

From an applied perspective, this research will benefit resource managers in the state of Vermont and beyond by advancing the knowledge of small watersheds in the Lake Champlain Basin. There are currently 17 small watersheds in Vermont that are considered impaired by stormwater runoff (VTDEC, 2004), and significant efforts are being undertaken by ANR to address this problem. Outside of Vermont, watershed managers and land use planners have become increasingly aware of the need to properly manage stormwater runoff and, in certain cases, restore highly-degraded streams in the urban environment (Brabec et al., 2002); yet the effectiveness of different mitigation approaches is not well reviewed and quantified (Bernhardt et al., 2005; Wohl et al., 2005). Mitigation of the hydrologic regime changes associated with urban development has been identified as a key component of urban watershed management (Booth and Jackson, 1997; Booth et al., 2004).

In Vermont, VTANR has proposed to mitigate the altered hydrologic regimes in the state's stormwater-impaired watersheds through the implementation of TMDL regulation using hydrologic targets. The results of my thesis research are being integrated into the permitting process that follows TMDL implementation to help identify priority areas for subwatershed scale mitigation. In addition, the comparison of urban and rural stream morphology (through the development of DHG regressions) will prove useful for future stream channel restoration efforts where reference conditions in the Lake Champlain Basin are needed as a basis for design.

Research Hypotheses

Given the gap in scientific knowledge and applied research identified through this literature review and summarized above, I developed and tested the following four hypotheses as part of my thesis research:

1) Geomorphic stability and macroinvertebrate richness declines as total impervious area (TIA) increases.
2) Upslope watershed TIA is the best predictor of stream condition (of any TIA scale measured).
3) Rural watersheds exhibit better defined downstream hydraulic geometry than urban watersheds.
4) Geomorphically-stable reaches support greater macroinvertebrate species richness than unstable reaches.

Figure 1. Sediment yield over time from as land is converted from agricultural to urban uses in the Piedmont of Maryland from Wolman (1967).

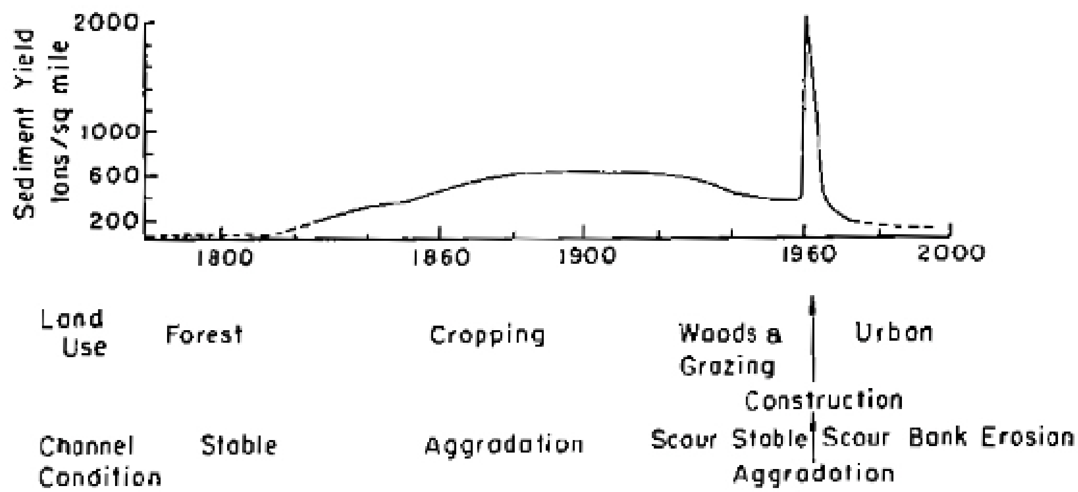
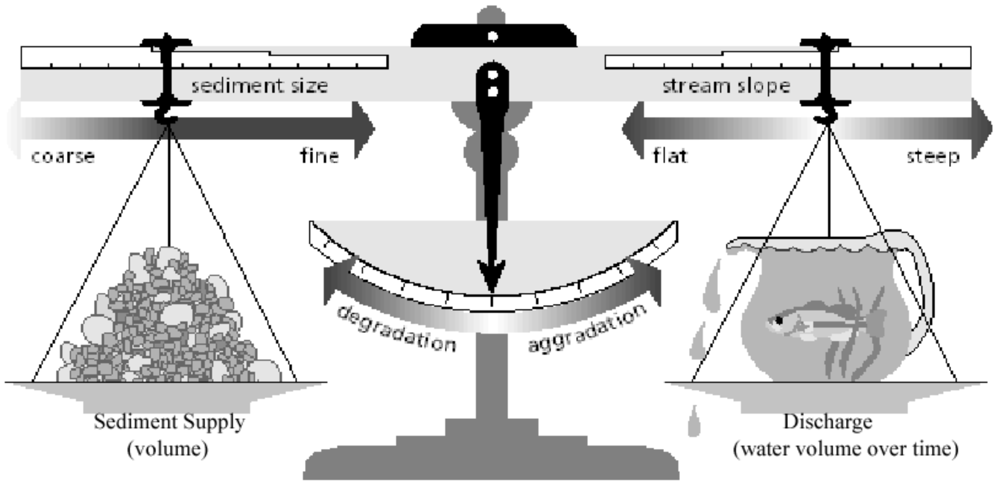


Figure 2. Lane's (1955) balance of sediment supply and sediment size with energy grade (slope) and discharge.



CHAPTER 2

LINKING URBANIZATION TO STREAM GEOMORPHOLOGY AND BIOTIC INTEGRITY IN THE LAKE CHAMPLAIN BASIN, VERMONT

Abstract

Impervious Cover Model (ICM) research has shown that a stream ecosystem response can be detected in urbanizing watersheds when the total impervious area (TIA) is at or above 10% of the watershed area, yet limited research has focused on the impacts of urbanization at different spatial and temporal scales. The physical and biotic conditions of stream reaches from 16 small watersheds in northwestern Vermont were assessed and analyzed for a response to total impervious area (TIA) at multiple spatial scales. Separate analyses were performed for high and low-gradient stream types, and reach selection criteria minimized the influence of human impacts to channel boundary conditions (e.g., bank armoring) to ensure a robust test of the ICM for upslope TIA alone. Response of geomorphic stability and sensitive macroinvertebrates to TIA was nonlinear and significant ($P < 0.001$), decreasing rapidly at 5% TIA. The effect of urbanization on stream condition interacts significantly with drainage area and channel slope ($P < 0.05$). Downstream Hydraulic Geometry (DHG) regressions developed for urban and rural watersheds and show a significant scale-dependent response ($p = 0.001$) of channel width to urbanization. Analysis of macroinvertebrate data from reaches in different stages of channel evolution indicates that stable reaches support greater EPT richness ($p < 0.005$) and overall species richness ($p < 0.01$) than unstable reaches. Results of ICM analyses and DHG regressions demonstrate that streams in Vermont may be more sensitive than those in other regions of the country in their response to urbanization, and that the response is scale-dependent. Results also indicate that some recovery of biotic communities may be possible following natural channel restabilization.

Introduction

The general mechanism for how urban land cover impacts stream ecosystems is well understood by watershed scientists. The conversion of natural, pervious surfaces to impervious cover (IC) leads to reduced rainfall infiltration through soils, causing an increase in peak discharges and a decrease in groundwater infiltration and stream baseflow (Leopold, 1968). The physical response of watershed processes and stream channel dynamics often leads to a decrease in drainage density as man-made drainage infrastructure replaces stream channels (Dunne and Leopold, 1978), an alteration of geomorphic structure and physical habitat of the channel (Wolman, 1967; Hammer, 1972; Booth, 1990; Pizzuto et al., 2000), an increase in pollutant conveyance to the channel (Roy et al., 2003; Coles et al., 2004), and a decline in biotic richness and diversity (Morely and Karr, 2002; Fitzpatrick et al., 2004; Miltner et al., 2004; Long and Schorr, 2005).

The Impervious Cover Model (ICM, Schueler, 1994) was developed to explain the general response of biotic and abiotic characteristics of stream ecosystems to IC impacts over a range of physiographic regions and has led to extensive testing of threshold effects in the urban environment. Much research has shown that a stream ecosystem response can be detected when the total impervious area (TIA) is at or above 10% of the watershed area (see review from CWP, 2003). The extensive body of literature that supports the ICM is useful for ecological researchers (Paul and Meyer, 2001) and land use planners (Brabec et al., 2002) interested in urban landscape management because it provides a direct causal relationship between urbanization and degradation of stream conditions. Given the public policy implications of the ICM (e.g., zoning regulations to control urbanization), there is considerable interest in whether the 10% TIA threshold reflects a true breakpoint in the health of stream ecosystems or whether there is a gradient of decline that is detectable with small amounts of watershed TIA below that level (CWP, 2003; Booth et al., 2004; Booth. 2005).

As noted by the CWP (2003), there may be regional differences in this response, as well as other land use impacts that confound the effect of TIA, including changes in forest cover (Booth et al., 2002) and legacy effects from previous land uses such as agriculture and forestry (Nelson and Booth, 2002). Recently published research indicates that impacts to stream ecosystems are detectable even when TIA is less than 10% in the Pacific Northwest (Booth et al., 2002), in the Central U.S. (Stepenuck et al., 2002; Wang and Kanhel, 2003) and in New England (Morse, 2001; Schiff and Benoit, 2007).

In addition to the ICM, downstream hydraulic geometry (DHG) regressions have proven useful for quantifying the physical response of stream channels to urbanization (Doll et al., 2002; Hession et al., 2003). DHG regressions were originally developed by Leopold and Maddock (1953) to describe changes in channel dimensions over a range of discharges, and have been promoted as tools for natural channel design in stream restoration projects (Rosgen, 1996). In self-formed alluvial channels, the DHG relationship for bankfull channel width is predicted using the following power equation:

$$W = \alpha Q^{\beta} \qquad (1)$$

where W is the bankfull channel width (m), Q is bankfull discharge (m^3/s), α is the regression coefficient, and β the regression exponent (and the slope of regression line). Bankfull channel depth and cross-sectional area can be substituted for width using the same power equation, and drainage area is often used as a surrogate for bankfull discharge in the absence of data for the latter. Although DHG regressions have been explored across different spatial scales (Wohl, 2004) and vegetation types (Anderson et al., 2004), relatively few DHG regressions have been developed and explored for urban watersheds to date (Allen and Narramore, 1985; Neller, 1989; Doll et al., 2002; Hession et al., 2003). DHG regressions that compare urban and rural watersheds from the same physiographic region may be useful in regional stream restoration efforts by aiding in quantification of urban impacts at different spatial scales,and in understanding variations in watershed processes along the channel network.

Greater understanding of the impacts of urbanization on stream ecosystems will be critical for improving land use management and policy in urban areas to protect water resources. Numerous studies have highlighted the importance of quantifying the impacts of urbanization on geomorphic condition, physical habitat (CWP, 2003) and aquatic biota at different spatial scales (Morley and Karr, 2002; McBride and Booth, 2005, Schiff and Benoit, 2007). This study explores these impacts in small watersheds in northwestern Vermont by testing: (1) the effect of TIA on geomorphic stability, physical habitat conditions, and biotic communities at three different spatial scales, (2) the differences between urban and rural DHG regressions, and (3) the response of biotic communities to different stages of urban channel evolution. We hypothesized that geomorphic stability and macroinvertebrate richness would decline as TIA increases, and that TIA measured at the watershed scale would better predict stream conditions than TIA measured at local scales. We also hypothesized that rural watersheds would exhibit better defined downstream hydraulic geometry than urban watersheds. Finally, we hypothesized that geomorphically-stable reaches would support greater macroinvertebrate species richness than unstable reaches.

Methods

Study Area

Seventeen small watersheds found within the Lake Champlain valley are listed as stormwater-impaired surface waters in need of a Total Maximum Daily Load Analysis (TMDL; VTDEC, 2004) and 9 of these are found in northwestern Vermont in the vicinity of the City of Burlington. As part of a larger effort to begin the TMDL process in Vermont's stormwater-impaired surface waters, the Vermont Agency of Natural Resources (VTANR) collaborated with the University of Vermont (UVM) and the Lake Champlain Committee (LCC) to collect reach-scale geomorphic stability and physical habitat data within 11 of the 17 stormwater-impaired watersheds. An additional 5 "attainment" watersheds that currently meet the VTANR water quality criteria were selected for study to provide a wide range of watershed TIA values and a contrast to the stormwater-impaired watersheds, making for a total of 16 study watersheds. Macroinvertebrate data collected by VTANR in 13 of the 16 study watersheds allowed for additional analysis of the effects of TIA and geomorphic stability on biotic integrity.

The 16 study watersheds are located in the Lake Champlain valley in northwestern Vermont (Figure 1). The topography of the valley is characterized by rolling terrain with mean watershed elevations ranging from 46 to 255 meters above mean sea level. The surficial geology is dominated by glacial till soils overlain by silts and clays deposited during the early Holocene when the entire valley was occupied by the Champlain Sea (Wright, 2003). Nine of the 11 stormwater-impaired watersheds are located in the vicinity of the City of Burlington, while two are in the City of St. Albans. The land use in the stormwater-impaired watersheds is characterized by a mix of residential, commercial, industrial, agricultural, and forested cover types representative of urbanizing watersheds. The attainment watersheds are located beyond the fringe of urban development surrounding the city of Burlington in rural landscapes with a mixture of forested and agricultural land uses with minimal amounts of low-density residential land use. Nearly all of the forested landscape in the Lake Champlain valley was cleared for agriculture during the mid to late 1800's and present day forested areas are second and third growth stands. Watershed TIA ranges from 0.6% to 39.3% across the study watersheds (Tables 1 and 2). Due to the higher degree of urban land cover found in stormwater-impaired watersheds, TIA values exceed 5% in these watersheds, while TIA values in the attainment watersheds are less than 5%. To simplify the terminology used in this study, stormwater-impaired and attainment watersheds are herein referred to as urban and rural, respectively.

Field Methods

The geomorphic stability and physical habitat conditions of stream reaches were surveyed by UVM and LCC during summer 2005 using a Stream Geomorphic Assessment (SGA) protocol developed by VTANR (VTDEC, 2005). Reaches were initially delineated using remote sensing techniques and a geographic information system (GIS) database that included surface water data, geology and soils data, and topographic mapping. Reach breaks were operationally defined along the channel network based on changes in (1) stream confinement (valley width), (2) valley slope, (3) geologic materials, and (4) tributary influences according to the VTANR SGA protocol (VTDEC, 2005). During the field data collection, reach delineations were verified or adjusted if necessary based on direct field observations of changes in valley and channel form and slope. In cases where further field segmentation was required due to changes in channel or valley characteristics that were not detectable with remotely sensed data, reaches were divided into smaller distances for surveying; however a minimum reach length of 100 m was maintained.

The SGA protocol utilizes a combination of quantitative and qualitative measurements to calculate two composite indices: (1) the Rapid Geomorphic Assessment (RGA) score; (2) the Rapid Habitat Assessment (RHA) score. Individual channel measurements used for RGA characterized the channel geometry and planform, bed substrate, bank erosion, and bank and buffer vegetation. At each reach, one to three cross-sectional measurements were taken at a representative riffle. For all reaches used in the development of DHG regressions, between three and four cross sectional measurements were collected at riffle locations where the thalweg cross-over occurs (VTDEC, 2005) to obtain average channel geometry dimensions. Bankfull width, floodprone width, mean depth, maximum depth, and low-bank height were collected at each cross section to calculate a bankfull width to depth ratio, entrenchment ratio, and incision ratio (Rosgen, 1996). Bed substrate was characterized by 100 randomly collected measurements across different geomorphic features in each reach using a pebble count methodology adapted from Wolman (1954) and described in the SGA protocol (VTDEC, 2005). This pebble count methodology resulted in values of the percentage of the substrate occupied by sands or finer (< 0.2 mm), fine gravel (2 -16 mm), coarse gravel (16 -64 mm), cobble (64 – 256 mm) and boulder (> 256 mm). Substrate embeddedness was evaluated during the sampling of bed substrate (Barbour et al., 1999). Large woody debris (LWD) that was at least 15 cm in diameter and 1.5 m in length was tallied for each reach. Sediment storage bars were evaluated for frequency and type (e.g., side bar). Stream bank material and cohesiveness was qualitatively evaluated, and the height and length of bank erosion and armoring was measured for each bank. Bank and buffer vegetation was classified based on general categories (e.g., coniferous, deciduous), and canopy cover (percent cover) was estimated for each bank.

Stream reaches were classified based on channel form using the Rosgen (1994) methodology, bed morphology and planform (Montgomery and Buffington, 1997), and channel evolution processes (Schumm et al., 1984; Thorne et al., 1997). The location of field survey points important for data analysis (e.g., cross-section locations) were recorded using a global positioning system (GPS). All field surveys were conducted during base flow conditions by observers trained during the same VTANR training session in May, 2005.

Using individual SGA field measurements described above, physical stream condition was quantified by assessing the effect of channel adjustment processes on reach equilibrium conditions (Lane, 1955). Assessed channel adjustment processes included: (1) degradation, (2) aggradation, (3) channel widening, and (4) planform change. The composite RGA calculation uses equal weighting of these four adjustment processes resulting in a score that ranges from 0 to 1.0, with a score of 1.0 reflecting a stable, undisturbed reach in dynamic equilibrium conditions (Lane, 1955). The RHA is a composite score adapted from the USEPA's Rapid Bioassessment Protocols (Barbour et al., 1999). The RHA combines individual quantitative measurements with a suite of qualitative observations to evaluate the following physical habitat conditions: epifaunal substrate and cover; pool substrate and variability; sediment deposition; channel flow status; channel armoring; channel sinuosity; bank stability and vegetation; and riparian buffer width. The composite RHA uses equal weighting of ten parameters to calculate a score that ranges from 0 to 1.0, with a score of 1.0 reflecting reference habitat conditions.

Macroinvertebrate sampling data collected from 13 of the 16 watersheds were obtained from VTANR's Biomonitoring Program. Samples were collected during September and October during baseflow conditions from representative riffles using a 500 μm mesh net (0.14 m^2 sampling area) following VTANR's composite riffle sampling methodology (VTDEC, 1989). Four separate locations within the riffle, representing a range of velocity and substrate types characteristic of that riffle, were sampled by agitating the substrate in a 0.2 m^2 area upstream of the mesh net. Each of the four locations were actively sampled for 30 seconds (total sample time of 2 minutes), and the contents collected in the mesh net were preserved in a container with 75% ethanol. Two replicate samples were collected from each riffle at independent locations within the riffle. In the laboratory, samples were thoroughly washed through 600 μm mesh brass sieve. The sample was then back-washed into a 30 cm x 45 cm tray delineated with 24 equally sized squares. A random selection technique was used to determine the order of square identification. All organisms were removed from each square before proceeding to the following selected square, and identification continued until a minimum of 6 squares (25% of the sample) had been completed. If less than 300 organisms had been identified in the first 6 squares, identification continued until a total of 300 organisms was reached. Organisms were identified to the species level by VTANR biologists, and tabulated species data were used to calculate the biotic index (BI; Hillsenhoff, 1987), sample richness and species richness for Ephemeroptera, Plecoptera, and Trichoptera (EPT) families. To explore linkages between current geomorphic stability (as assessed in 2005) and biotic integrity, we used only VTANR macroinvertebrate sampling results collected during the period of 2000 to 2005. Due to substantial urbanization within many of the study watersheds during the past ten years, samples collected prior to 2000 might reflect biotic communities found under different hydro-geomorphic conditions, and were therefore omitted. For the sampled reaches used in statistical analyses, between one and six macroinvertebrate samples were collected for each reach (an average of two per reach) over this time period.

Spatial Analysis of TIA

GIS software (ESRI ArcGIS®) was used to delineate and quantify watershed areas and land use at three different spatial scales: (1) the complete upslope drainage area (upslope area) to the reach, (2) the local drainage area (local area) to the reach, and (3) the stream corridor (corridor) defined as the area within 30 m of either side of the stream channel (Figure 2). For statistical analyses using the percent TIA of the upslope area as one independent variable, TIA was measured for the drainage area beginning at the downstream reach break and extending upslope.

Land use data derived from two separate sources for the study area was utilized to quantify percent TIA for each drainage area. Statewide Landsat imagery collected in 2002 using a 30 m grid was processed by UVM's Spatial Analysis Laboratory (SAL), resulting in the following four spectral classes: (1) forest; (2) urban; (3) open (agricultural and open recreational uses); (4) water and other (SAL, 2005). In addition, a separate dataset of TIA derived from high-resolution Quickbird satellite scenes collected between 2003 and 2005 was utilized (Morrissey and Pelletier, 2006). The multispectral bands (2.4 m resolution) from the Quickbird scenes were analyzed by SAL using Definiens eCognition® software to classify the data into three classes: (1) impervious; (2) pervious; (3) water. Quickbird-derived TIA data was only available for a select group of watersheds during the time of this analysis, whereas the Landsat-derived urban coverages were available for the entire study area. Given this limitation, a correlation analysis was performed using the Landsat-derived urban class and the Quickbird-derived TIA class that resulted in a robust linear relationship (R^2 = 0.96) that was used to calculate percent

TIA for all study watersheds at each spatial scale using the Landsat urban class as the predictor variable. The methods used to develop this relationship are described in further detail in Appendix B.

Reach Selection Criteria

To test the ICM in our study area, we chose to utilize an independent watershed approach to measure the effect of TIA on field measured variables of stream condition. This approach to site selection, similar to that taken by other researchers (Roy et al., 2003; Wang and Kanhel, 2003; Coles et al. 2004), results in a single site per watershed to test for land use effects on completely independent reaches. Two of the 16 study watersheds contained more than one independent subwatershed, resulting in a total of 18 possible locations for reach selection. Using this approach, we selected reaches from each independent subwatershed for the following two geomorphic stream types: (1) high-gradient; (2) low-gradient. The criteria used for determining stream type are consistent with VTANR's SGA protocols for the calculation of RGA and RHA scores and are based primarily on channel form (Rosgen, 1994), bed substrate, and channel slope. High-gradient reaches are characterized by coarse-grained bed substrate with channel slopes greater than 0.5%, and low-gradient reaches are characterized by fine-grained bed substrate with channel slopes less than or equal to 0.5%. Separate ICM analyses of high and low-gradient stream types were conducted to reduce the gradient of natural characteristics which covary with anthropogenic factors (Allan, 2004). For both stream types, reaches were screened for selection using the following additional criteria: (1) the reach is located in the most downstream area of the watershed; (2) the stream corridor has limited legacy effects of channel straightening, berming or armoring. The first criteria aids in selecting larger sized (e.g., width) stream reaches having features that are more readily observed than in lower-order (Strahler, 1964) stream reaches. The second criteria screens reaches with direct impacts to channel boundary conditions, thereby allowing for a more specific ICM test of the effect of upslope TIA alone. Using these criteria within the 18 independent watersheds, 17 reaches were suitable for use in the high-gradient analysis (Table 1) and twelve reaches were suitable for use in the low-gradient analysis (Table 2). Fewer low-gradient reaches were available because some of the smaller watersheds had high relief ratios (>30; Dunne and Leopold, 1978) with no low-gradient reaches. Macroinvertebrate samples were available only in the high-gradient reaches where coarser bed substrates are suitable for VTANR biomonitoring protocols (VTDEC, 1989). Therefore, the analysis of biotic response to TIA was limited to the high-gradient dataset (Table 1).

Reaches selected for DHG regression analysis included multiple sites per watershed. This approach is consistent with the original DHG methods developed by Leopold and Maddock (1953) and those used in recent analyses in small urban and rural watersheds (Doll et al., 2002; Hession et al., 2003). Reaches were classified as non-alluvial channels and were eliminated if they contained significant bank or bed armoring or were located in close proximity to bridges or culverts where human-caused adjustments are prevalent. Separate DHG regressions were developed for high and low-gradient stream types.

Reaches used in exploring linkages between geomorphic stability and macroinvertebrate sampling data also included multiple sites per watershed. For this analysis, the reach selection criteria for completely independent reaches was considered less important since the stressor and response are localized at the reach scale, unlike the ICM test of the effects of upslope watershed TIA. Twenty-six high-gradient reaches within 13 of the 18 independent subwatersheds with available macroinvertebrate and geomorphic assessment data were selected for this analysis.

Statistical Analysis

For all ICM analyses, variables with non-normal distributions were log transformed to normalize the data for analysis using MINITAB® Statistical Software. Correlation analysis on ranked data was used to produce a matrix of Spearman's ρ values, which allowed for exploration of collinearity between variables. Single predictor models that analyzed the effect of TIA on physical stream conditions and biota were developed using least squares regression on log transformed data. A stepwise regression analysis (backward elimination; $P = 0.1$ to remove) was then performed using TIA and other natural watershed characteristics to determine the best-fit multiple linear regression models for each spatial scale of TIA and to determine the relative influence of each predictor variable on stream condition. Due to the small sample sizes for ICM analyses of independent reaches for physical conditions ($N = 17$) and biotic conditions ($N = 14$), multiple linear regression models were restricted to three variables with no interaction terms. However more complicated effects involving the interaction of urbanization with watershed drainage area and channel slope were explored using analysis of covariance (ANCOVA).

DHG regressions were developed using the power law equation approach (Leopold and Maddock, 1953; Hession et al., 2003). Data for low-gradient reaches yielded poorly developed DHG relationships (Wohl, 2004) and were therefore not analyzed in the same detail as the regressions for high-gradient reaches. DHG regressions for high-gradient reaches were developed using 24 reaches from urban watersheds and 19 reaches from rural watersheds (Table 3). ANCOVA was used on log transformed data for high-gradient reaches to test for interaction of watershed drainage area (DA) with the urbanization effect (watershed type) in the response of channel dimensions, following Hession et al. (2003). The ANCOVA approach tested for significant DHG regressions and for evidence of a scale-dependent response of channel dimensions to urbanization.

Boxplots depicting the response of macroinvertebrates to channel evolution stages for incising reaches (Schumm et al., 1984; VTDEC, 2005) were visually evaluated for trends. Channel evolution stages were divided into stable and unstable categories (VTDEC, 2005) and a non-parametric Mann-Whitney test of the population medians was performed on the two categories for EPT richness and overall species richness.

Results

ICM Analysis

Correlations of ranked data (Table 4) showed that RGA and RHA were highly correlated and therefore a single metric, RGA, was used to summarize ICM results for physical stream conditions. Similarly, EPT richness and overall species richness were significantly correlated and EPT richness was chosen to summarize ICM results for biota. Upslope percent TIA and percent forest cover were significantly correlated, and because the purpose of this study was to determine the effect of urbanization on stream conditions, percent forest cover was excluded from the regression analyses. Channel slope and watershed DA, variables that are typically correlated in many watersheds as slope decreases in a downstream direction across watershed zones (Dunne and Leopold, 1978), were also significantly negatively correlated in our study area. Percent cobble substrate and sand substrate were also significantly negatively correlated.

Single predictor regressions using TIA as a percentage of the upslope area resulted in significant non-linear models for predicting RGA (Figure 3) and EPT richness (Figure 4) in high-gradient reaches. The single predictor model for RGA was significantly better for high-gradient reaches (Figure 3) than the model for low-gradient reaches (Figure 4). The comparison of these two models show a steeper declining response in high-gradient RGA (Figure 3; regression slope = -0.15) than in low-gradient RGA (Figure 5; regression slope = -0.04) in response to TIA. A threshold of decline at 5% TIA was observed for high-gradient reaches in the response of geomorphic stability, as measured by RGA (Figure 3); and in biota, as measured by EPT richness (Figure 5).

Above 10% TIA, the response to TIA was not significant for RGA (N = 6, $P = 0.426$) or EPT (N = 6, $P = 0.157$).

Stepwise regression analyses for multiple spatial scales showed that TIA was a significant variable in all models of RGA (Table 5). For the upslope and local area models, TIA was the only significant variable in the model, however DA was also a significant variable ($P < 0.1$) in the corridor model. TIA was a significant variable in upslope and local area models of EPT, but not the corridor model (Table 5). Percent cobble substrate was a significant variable and explained 11% of the variance of EPT in the upslope area model (Table 5). For the corridor model of EPT, only DA and percent sand substrate were significant predictors. Reach elevation and channel slope were not significant in any of the models developed through the stepwise regression analysis.

Although channel slope was not a significant variable in the stepwise regression analysis, ANCOVA results showed that both channel slope and DA were significant covariates ($P < 0.05$) in the upslope area model of RGA (Table 6; Figure 6); confirming that variables associated with watershed scale interact with urbanization in its effect on physical stream condition. Neither channel slope nor DA were found to be significant covariates in the EPT models.

DHG Regressions

Using bankfull geometry measurements for high-gradient streams (Table 3), DHG regressions were developed for the response of channel dimensions to the effect of DA (Figure 7). Regressions for channel width, depth and cross-sectional area (variable not plotted in Figure 7) were all significant (Table 6) and had R^2 values greater than 0.50, indicating well developed DHG relationships (Wohl, 2004) for high-gradient reaches. Using ANCOVA, we found a significant effect of urbanization and a significant urbanization-DA interaction for channel width, and for channel cross-sectional area but not for channel depth (Table 6). Results indicate that there is a scale-dependent response of channel width and cross-sectional area to urban land cover. This effect is more pronounced in lower-order reaches and becomes insignificant at a DA of ~15 km^2.

Biotic Response to Channel Stability

Geomorphic assessment data and macroinvertebrate sampling data were analyzed for 26 high-gradient stream reaches to explore the response of biota to channel stability. A box and whisker plot depicting the response of EPT richness to channel evolution processes (Figure 8) revealed a pattern of decline beginning with the incision and widening stages (II and III), and recovery through the aggradation (IV) and restabilization (V) stages. A similar trend was noted for the response of overall species richness. Reaches classified as stages I (equilibrium conditions) and V (quasi-equilibrium conditions) were grouped as stable ($N = 4$), and those classified as stages II through IV were grouped as unstable ($N = 22$). Results of Mann-Whitney tests of the population medians indicate that stable reaches support greater EPT richness ($P < 0.005$) and overall species richness ($P < 0.01$) than unstable reaches.

Discussion

ICM Thresholds

There is considerable evidence supporting a range of TIA threshold values between 10%-20% (see reviews from CWP, 2003; Allan, 2004). However natural gradients and legacy effects may confound the relationship, making it unlikely that a single threshold can be applied to all regions (Allan, 2004). The CWP (2003) notes that the ICM has been tested in the cold climate regions of the upper Midwestern U.S. and New England, but it appears that little or no exploration of regional threshold differences have been conducted to date. Our results using high-gradient reaches add to the growing number of results indicating that stream conditions decline rapidly at levels below 10% TIA in the upper Midwestern U.S. and New England (Stepenuck et al., 2002; Wang and Kanhel, 2003; Morse, 2005; Schiff and Benoit, 2007) and may respond differently from other regions of the U.S. Our nonlinear results for high-gradient reaches (Figures 3 and 5) are very similar to those observed in Maine (Morse, 2001) and Connecticut (Schiff and Benoit, 2007). These relationships are characterized by some variability in biotic integrity and geomorphic stability below 5% TIA, with a sharp decline between 5%-10% TIA, followed by a leveled response in watersheds with TIA greater than 10%. The variability in stream ecosystem response below 5% TIA in New England is likely attributable to other natural variables (Booth and Jackson, 1997) and stream reaches in this ICM setting are generally termed "sensitive" (CWP, 2003).

Our results contrast with the general findings of the CWP (2003) as to where the response gradient from "sensitive" to "impacted" occurs. The ICM suggests that this shift is generally observed at 10% and that greater variability in stream health is often observed below this level. The results of this study, along with those of Morse (2001) and Schiff and Benoit (2007), indicate that streams in New England may exhibit this variability at levels as low as 5% TIA and perhaps lower. Our high-gradient reach models show a similar, nonlinear response for physical channel conditions (RGA; Figure 3) and sensitive biota (EPT richness; Figure 5), providing strong evidence for the trend using two separate measurements of stream ecosystem health. We feel that the independent reach approach to ICM testing was an important component of our study and may explain why we observed very strong relationships for high-gradient reaches.

We found that reaches with greater than 10% upslope TIA did not significantly respond to TIA in terms of physical condition or sensitive biota. Similar results from other stream studies in the northeastern U.S. have been reported for various biotic (Morse, 2001), physical (Cianfrani et al., 2006) and water quality indices (Schiff and Benoit 2007) at levels above 10% TIA. Cianfrani et al. (2006) analyzed an extensive dataset of physical channel conditions from Pennsylvania to test the ICM, and reported that 6 out of 10 response variables were significantly correlated with TIA in watersheds with less than 13% TIA, whereas only 2 out of 10 response variables were significantly correlated with TIA in watersheds with greater than 24% TIA. These results from Cianfrani et al. (2006) provide further evidence that stream conditions in their study region may respond more severely at lower levels of urbanization than in other regions of the U.S. Taken together, the results from the northeastern U.S. indicate that further increases in watershed TIA above 10% may not result in significant additional decline in stream ecosystem health.

Why there appear to be differences in the response of stream condition to urbanization between regions is unclear. Fitzpatrick et al. (2005) noted that the differences in this response may be related to the legacy effects of previous land uses. In the upper Midwestern U.S. and New England, nearly the entire landscape has a history of multiple human impacts and the sensitivity of watersheds to further human impacts may be heightened. In these regions, where impacts to the infiltration capacity (e.g., reduced organic matter) of the soils have occurred and recurred over the last 200 years due to agriculture and forestry practices, it is possible that minor impacts from urbanization could lead to significant declines in stream condition. The corollary may also be true for those regions in the Pacific Northwest where urban land cover is replacing forests with little history of disturbance. Further investigation is required to determine why regional patterns of stream ecosystem response to urbanization exist.

Natural Gradients

Given the independent watershed approach and reach selection criteria used in this analysis, our results provide clear evidence that stream ecosystem health declines in response to TIA in our study area. However, we also observed very different physical responses between high and low-gradient stream types. The decline of geomorphic stability in response to TIA was more significant and rapid in high-gradient reaches than in low-gradient reaches. Differences in natural factors between the two stream types such as channel slope, floodplain connectivity and channel boundary conditions (e.g., stream bank soil and vegetation) may be important in explaining the differences in response to TIA. Low-gradient stream types in our study area tend to have wider floodplains, greater sinuosity, and herbaceous stream bank vegetation. Greater dissipation of the erosive peak flow forces caused by urbanization may be achieved by low-gradient streams that have adequate floodplain access and the ability to develop sinuosity more readily than high-gradient streams. In addition, many studies have shown that stream channels with herbaceous bank vegetation tend to be narrower than those with woody vegetation (Davies-Colley, 1997; Hession et al., 2003; Anderson et al., 2004), a difference which may result from lower rates of erosion in channels with the former condition (Trimble, 1997). These factors may be responsible for the trend observed in the response of RGA to TIA, whereby each percentage increase of watershed TIA appears to impact physical stream conditions less in low-gradient reaches than in high-gradient reaches (Figures 3 and 6).

The presence of beavers (*Castor canadensis)* in many of the low-gradient reaches in the study area made the assessment of physical channel conditions more difficult and may have introduced additional variability in our analysis. Through dam building and tree removal, beavers drastically impact the hydrologic and geomorphic characteristics of streams in our region (Naiman et al., 1986; Naiman et al., 1994). For ICM studies conducted in regions where beavers are abundant in low-gradient stream types, TIA prediction of stream conditions may be improved by separating stream types based on gradient as we have done in our study.

Our high-gradient stream results indicate that the response of stream condition depends on the interaction of other natural or inherent watershed characteristics with urban land cover. In our urban study watersheds, steep headwater reaches appear to be more impacted by the effects of urban land cover than downstream, higher-order reaches. Other studies have also reported that steep headwaters channels are most susceptible to rapid destabilization (Neller, 1989; Booth, 1990). Neller's (1989) research in Australia found that the magnitude of urban-induced channel enlargement in steeper, headwaters reaches was four to five times greater than lesser sloped, downstream reaches. In urbanizing watersheds in the Pacific Northwest, Booth (1990) noted that lower-order, higher-gradient stream reaches are most susceptible to rapid channel incision processes, and that high sheer stresses in low-gradient, higher-order streams produced only minimal bed lowering. Channel incision processes are typically followed by bank failure and channel widening and lower-order channels may be progressing more rapidly through channel evolution stages than higher-order channels.

Scale-Dependent Responses

Exploration of the effect of localized TIA on stream condition revealed a slightly different pattern than the upslope TIA analysis. Models developed to predict RGA and EPT richness using percent TIA in the local area were also significant, but explained less variance than upslope area models. However, unlike upslope area models, local area and corridor models of EPT were improved slightly when the DA variable was included with TIA (Table 5). This result suggests that while localized TIA may also have an effect on stream condition, this effect depends on the size of the entire watershed area draining to the reach. This effect would be obvious in the case of a high-order reach located in the downstream area of a rural watershed. In this case, higher TIA in the local area may not have a significant effect on stream conditions, as its effect would become dampened by the natural watershed conditions upslope. Many studies have found significant correlations between local and upslope urban land cover (Fitzpatrick et al., 2001; Morley and Karr, 2002; Wang and Kanehl, 2003; McBride and Booth, 2005; Schiff and Benoit, 2007), making it difficult to conclude which scales are important for management of TIA and its effects on stream conditions. In our study, percent TIA in the local area was highly correlated with percent TIA in the upslope area ($\rho = 0.87$, $P < 0.001$), as were corridor and upslope area TIA ($\rho = 0.63$, $P = 0.006$). Therefore strong conclusions cannot be drawn about the importance of measuring localized TIA. However, our results generally agree with other urban watershed studies showing that localized and upslope urban land cover may both be important variables in predicting stream condition (Wang et al., 2001; Morley and Karr, 2002; Wang and Kanehl, 2003; McBride and Booth, 2005). In addition, our results indicate that the predictability of stream condition declines as TIA is measured at smaller spatial scales within the watershed.

Results of the DHG regressions for high-gradient stream types provide strong evidence of a scale-dependent response of channel width to urbanization (Figure 6). In reaches where DA is small ($1 km^2 – 5 km^2$), urban stream channels are wider than rural channels; however this response diminishes as the DA approaches ~15 km^2. The same response was observed for channel cross-sectional area, but not for channel depth. We explored possible explanations for the width and cross-sectional area response, including correlations between DA and TIA, and differences in channel slope between urban and rural stream types. We found no significant correlation between DA and TIA (Table 4), and no significant difference between the median value of the slope populations for the two stream types ($\alpha = 0.05$).

Numerous other studies of DHG regressions in small urban and rural watersheds show a similar response (Allen and Narramore, 1985; Neller, 1989; Doll et al., 2002;

Hession et al., 2003). Neller (1989) showed that urban channel cross-sectional areas are systematically larger than their rural counterparts, and that this effect is magnified at smaller drainage areas in Australia. Allen and Narramore (1985) showed this same effect with channel width in two different lithologies, shale and chalk-bottomed streams. Hession et al. (2003) studied rural and urban streams in the Mid-Atlantic Piedmont region, detailing the differences in hydraulic geometry between the two stream types and their bank vegetation types. Results from Hession et al. (2003) also show that the slopes and intercepts of the DHG regressions were significantly different between the two land use settings in both forested and nonforested environments. Doll et al. (2002) also found that urban streams are significantly wider and larger in cross-sectional area by comparing channel geometry in a study of urban and rural streams in the Piedmont region of North Carolina.

Biotic Response to Channel Stability

Our results of macroinvertebrate response to channel evolution processes suggest that there is a link between geomorphic stability and biotic communities in small urban watersheds (Figure 8). Excess stream power and sheer stresses associated with the incision and widening stages (II and III) are likely leading to more frequent scour of the stream bed, and a subsequent loss of the quality habitat needed to support sensitive macroinvertebrate taxa. Stream power typically decreases as channels aggrade coarse material and redevelop sinuosity in stage IV of channel evolution, and stream power in the quasi-equilibrium channel stage (V) is often similar to that observed in stable channels (Bledsoe et al., 2002). Other studies in Vermont have linked geomorphic processes with biotic integrity in rural watersheds (Sullivan et al., 2004; Sullivan et al., 2006), however these results have been less conclusive for macroinvertebrates than fishes. Sullivan et al. (2004) carried out a study using RGA, RHA and macroinvertebrate measurements from paired stable and unstable reaches in central Vermont. Although Sullivan et al. (2004) reported that stable reaches did not support significantly greater macroinvertebrate densities than unstable reaches, EPT richness was significantly correlated with geomorphic stability. Sullivan et al. (2006) also reported on the influence of geomorphic condition on fish communities in separate study sites in Vermont. Results from this study indicate that geomorphic stability was a significant predictor of fish communities, and suggest that fishes may be more responsive to geomorphic stability than macroinvertebrates due to differences in habitat scales.

A growing body of literature suggests that the maintenance of ecological integrity (Angermeier and Karr, 1994) in riverine ecosystems involves the complex interaction of many physical and biological processes at multiple spatial scales (Thompson et al., 2001; Poole, 2002; Richards et al., 2002; Aarts et al., 2004). The physical processes involving a balance between water and sediment inputs (Lane, 1955) that occur in river corridors are inherently dynamic (Schumm, 1977), and these multi-scaled processes create and maintain the heterogeneity in physical habitat which in turn supports diverse biotic communities. In the urbanized watersheds of our study area, excess hydrologic loading is the dominant stressor and results in sediment export and channel incision. Once the channel incision process has begun, the loss of channel sinuosity reduces the potential surface area of habitat suitable for macroinvertebrate colonization. Narrow, incised channels with homogeneous habitat appear to support less diverse biotic communities until channels begin to redevelop sinuosity during their evolution into a new state of quasi-equilibrium (Figure 8; Stage V). The typical response of biota to channel evolution processes in urban watersheds may be different from that observed in rural watersheds where other adjustment processes (e.g., aggradation) are more common. These differences in physical adjustment processes may explain why the relationships between geomorphic stability and biotic richness we report are stronger than those reported by Sullivan et al. (2004) from the same region.

Summary and Conclusions

This study provides additional evidence that relatively low levels of urbanization negatively impacts stream conditions in the northeastern U.S., and the response to this stressor can be measured using physical and biotic indices. The response of stream condition to urbanization was significant for all scales at which TIA was measured. Other studies have suggested that the relationships between stream condition and urbanization may be regionally dependent (Cianfrani et al., 2006). Further exploration of this response across regions of the U.S. would be useful for aquatic ecologists and land use planners working on watershed and stream restoration problems.

In Vermont, the results of this study have implications for how local government (e.g., town planning commission) could address planning and zoning to avoid further degradation of water resources. Vermont is largely a rural state, but the Burlington area has seen rapid urbanization during the past 20 years in response to a growing local economy. Many towns and cities in the northwestern part of the state have responded by redeveloping zoning plans to accommodate this growth, and some municipalities are considering the implementation of a “stormwater utility” to manage urban runoff in the stormwater-impaired watersheds found within their municipal boundaries. Further recognition of the sensitivity of small streams to low levels of urbanization would help planners understand and protect valuable water resources in the Lake Champlain Basin.

The results of this study also highlight the importance of considering watershed scale and other inherent characteristics in the response of stream conditions to urbanization. Headwaters areas with steep channel slopes are the most impacted zones in small urban watersheds in northwestern Vermont. This information will aid VTANR in its development of TMDL restoration plans in the stormwater-impaired watersheds. DHG regressions comparing urban and rural watersheds proved useful in elucidating the scale-dependent response in our study area, and this approach could be useful in other areas where similar studies are in progress.

Lastly, this study provides insight into the influence of channel stability on aquatic biota in urban watersheds. Despite the abundance of data supporting the hypothesis that urbanization negatively affects stream ecosystems, fewer studies have explored whether biotic communities recover following natural or human-induced channel restoration in urban watersheds. Answers to this challenging question will become critically important as more resources are invested in urban channel restoration in the U.S. Our results provide evidence that some recovery may be possible following natural restabilization; or perhaps engineered restoration which mimics this natural process. However Booth (2005) notes that there is little evidence suggesting that the mitigation of nonhydrologic factors (e.g., channel geometry) can remediate the consequences of urbanization. Channel evolution processes, and the response of aquatic biota to them, are directly tied to the altered sediment and hydrologic regimes brought on by urbanization. Therefore, remediation efforts in urban-impaired watersheds will need to address the altered hydrologic regime at the watershed scale *and* provide the lateral space within the stream corridor for streams to restabilize (Bernhardt and Palmer, 2007). As noted by Wohl et al. (2005), future watershed and channel restoration efforts will be greatly enhanced by additional research which adds to our knowledge of these watershed-scale process dynamics.

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Table 1. Characteristics of high-gradient study reaches

Watershed Name	Watershed Type	Drainage Area (km^2)	TIA	Channel Slope	Elevation (m)	Bed Substrate#	Channel Bedform*	Stream Type**	RGA Score	RHA Score	Biota Sample N§	EPT Richness†	Overall Richness††
Allen Bk (lower)	Urban	29.0	7.0%	1.0%	63	fine gravel	Pool riffle	C	0.56	0.60	1	6	36
Bartlett Bk	Urban	2.7	15.2%	3.2%	33	fine gravel	Step pool	B	0.48	0.53	2	6	24
Centennial Bk	Urban	3.9	29.0%	3.3%	53	coarse gravel	Step pool	B	0.53	0.59	3	3	28
Englesby Bk	Urban	1.9	21.0%	2.1%	46	fine gravel	Plane bed	B	0.38	0.44	1	3	31
Indian Bk	Urban	19.5	9.0%	2.4%	75	fine gravel	Plane bed	C	0.58	0.64	2	10	42
Morehouse Bk	Urban	1.1	32.6%	4.9%	64	coarse gravel	Plane bed	B	0.28	0.47	2	5	25
Munroe Bk (main stem)	Urban	6.3	5.7%	1.3%	59	fine gravel	Pool riffle	C	0.59	0.68	1	9	22
Munroe Bk (tributary)	Urban	3.2	5.9%	1.7%	51	coarse gravel	Plane bed	C	0.55	0.46	2	8	26
Potash Bk	Urban	18.1	19.8%	1.2%	39	coarse gravel	Pool riffle	C	0.45	0.37	6	7	28
Rugg Bk	Urban	6.8	8.1%	1.7%	127	fine gravel	Pool riffle	C	0.63	0.65	2	7	34
Stevens Bk	Urban	4.0	10.8%	3.2%	128	fine gravel	Step pool	B	0.45	0.54	1	8	44
Allen Bk (upper)	Rural	10.1	2.7%	2.9%	170	cobble	Plane bed	B	0.78	0.85	1	25	68
Johnnie Bk	Rural	16.6	1.2%	3.6%	99	cobble	Step pool	B	0.85	0.88	N.A.	N.A.	N.A.
Mill Bk	Rural	41.9	1.6%	2.8%	98	cobble	Plane bed	B	0.75	0.80	2	27	64
Streeter Bk	Rural	17.3	3.9%	1.4%	63	fine gravel	Pool riffle	C	0.78	0.90	3	14	49
Sucker Bk	Rural	17.7	2.6%	1.3%	108	coarse gravel	Pool riffle	C	0.70	0.69	N.A.	N.A.	N.A.
Trout Bk	Rural	1.9	3.1%	4.5%	78	cobble	Step pool	B	0.85	0.84	N.A.	N.A.	N.A.

Median bed substrate: fine gravel (2-16 mm), coarse gravel (16-64 mm), and cobble (64-256 mm)

* From Montgomery and Buffington (1997)

** From Rosgen (1994)

§ Number of macroinvertabrate samples used for average richness values

† Species richness for Ephemeroptera, Plecoptera, and Trichoptera

†† Overall species richness for entire sample

N.A. Macroinvertebrate data not available for reach

Table 2. Characteristics of low-gradient study reaches

Watershed Name	Watershed Name	Drainage Area (km^2)	TIA	Channel Slope	Elevation (m)	Bed Substrate#	Channel Bedform*	RGA Score	RHA Score
Allen Bk	Urban	18.6	4.1%	0.35%	134	sand	Plane bed	0.54	0.62
Centennial Bk	Urban	3.8	31.1%	0.46%	65	sand	Dune ripple	0.59	0.57
Indian Bk	Urban	31.3	8.4%	0.23%	29	sand	Dune ripple	0.73	0.71
Morehouse Bk	Urban	0.7	39.3%	0.47%	72	sand	Dune ripple	0.53	0.37
Munroe Bk (main stem)	Urban	9.2	7.0%	0.48%	50	sand	Plane bed	0.40	0.42
Munroe Bk (tributary)	Urban	4.6	4.9%	0.42%	46	sand	Dune ripple	0.68	0.58
Potash Bk	Urban	15.4	20.3%	0.41%	65	sand	Dune ripple	0.59	0.66
Sunderland Bk	Urban	10.5	16.1%	0.16%	43	sand	Dune ripple	0.56	0.50
Johnnie Bk	Rural	17.0	1.1%	0.48%	90	fine gravel	Pool riffle	0.64	0.56
Mill Bk	Rural	18.6	0.6%	0.31%	236	fine gravel	Pool riffle	0.73	0.70
Sucker Bk	Rural	18.7	2.6%	0.20%	103	sand	Plane bed	0.68	0.69
Trout Bk	Rural	12.2	1.7%	0.34%	31	sand	Plane bed	0.68	0.57

§ Median bed substrate: sand or finer (< 0.2mm); fine gravel (2-16 mm)

* From Montgomery and Buffington (1997)

Note: all reaches classified as type E per Rosgen (1994)

Table 3. Data summary for high-gradient reaches used to develop DHG regressions

Watershed	Watershed Type	Reach	Drainage Area (km^2)	Channel Width (m)*	Channel Depth (m)*	Channel Area (m^2)*
Allen Brook	Urban	1	29.1	9.5	0.8	7.6
Allen Brook	Urban	2	27.9	11.8	0.8	9.5
Allen Brook	Urban	3	26.3	9.8	0.6	5.8
Allen Brook	Urban	4	26.3	8.6	0.5	4.2
Allen Brook	Urban	5	13.5	5.8	0.4	2.2
Bartlett Brook	Urban	1	2.7	3.6	0.4	1.5
Bartlett Brook	Urban	2	1.7	4.0	0.5	1.8
Bartlett Brook	Urban	3	1.3	3.8	0.2	0.8
Centennial Brook	Urban	1	4.0	6.4	0.5	3.2
Centennial Brook	Urban	2	1.1	6.0	0.3	1.8
Englesby Brook	Urban	1	1.9	7.0	0.3	2.4
Indian Brook	Urban	1	19.6	8.6	0.4	3.1
Indian Brook	Urban	2	13.9	7.0	0.6	3.9
Morehouse Brook	Urban	1	1.1	4.3	0.2	0.8
Munroe Brook	Urban	1	13.9	6.1	0.4	2.5
Munroe Brook	Urban	2	6.1	8.4	0.5	4.1
Munroe Brook	Urban	3	3.7	4.5	0.2	1.0
Munroe Brook	Urban	4	4.6	3.5	0.3	1.0
Potash Brook	Urban	1	18.4	10.5	0.5	4.9
Potash Brook	Urban	2	18.2	8.5	0.5	4.5
Potash Brook	Urban	3	15.6	10.0	0.6	5.5
Potash Brook	Urban	4	11.3	6.7	0.4	2.7
Rugg Brook	Urban	1	7.4	4.8	0.3	1.5
Rugg Brook	Urban	2	6.8	6.3	0.4	2.3
Allen Brook	Rural	6	10.2	4.9	0.3	1.7
Allen Brook	Rural	7	4.1	4.0	0.3	1.3
Johnnie Brook	Rural	1	16.6	9.9	0.4	4.3
Johnnie Brook	Rural	2	16.4	10.9	0.5	5.9
Johnnie Brook	Rural	3	14.8	9.5	0.5	4.6
Johnnie Brook	Rural	4	10.0	4.7	0.3	1.4
Johnnie Brook	Rural	5	9.5	8.4	0.3	2.8
Johnnie Brook	Rural	6	6.0	4.8	0.5	2.2
Johnnie Brook	Rural	7	3.0	3.6	0.4	1.4
Johnnie Brook	Rural	8	1.3	2.5	0.3	0.7
Mill Brook	Rural	1	42.0	13.6	0.6	8.1
Mill Brook	Rural	2	36.3	16.9	0.6	9.9
Mill Brook	Rural	3	36.0	10.6	0.6	5.9
Mill Brook	Rural	4	35.1	11.7	0.5	5.8
Mill Brook	Rural	4	32.5	13.3	0.5	6.9
Mill Brook	Rural	5	30.8	10.7	0.6	6.6
Mill Brook	Rural	6	14.2	7.8	0.5	4.0
Sucker Brook	Rural	1	17.8	6.4	0.4	2.6
Sucker Brook	Rural	2	16.9	5.5	0.5	2.6

* Average of 3 to 4 bankfull values from each reach

Table 4. Spearman's rank correlations for landscape variables and stream physical and biotic indices.

	DA[†]	TIA[#]	Forest[##]	Slope[††]	% Cobble	% Sand	RGA	RHA	Richness[§]	EPT[§]	BI[§]
DA[†]	1										
TIA[#]	-0.48	1									
Forest[##]	0.67 **	-0.80 ***	1								
Slope[††]	-0.56 *	0.12	-0.02	1							
% Cobble	0.03	-0.32	0.35	0.53 *	1						
% Sand	-0.27	0.26	-0.60 *	-0.36	-0.74 **	1					
RGA	0.40	-0.90 ***	0.75 **	0.03	0.38	-0.48 *	1				
RHA	0.46	-0.79 ***	0.69 **	-0.01	0.44	-0.43	0.93 ***	1			
Richness[§]	0.64 *	-0.53	0.83 ***	-0.04	0.36	-0.58 *	0.55 *	0.45 *	1		
EPT[§]	0.63 *	-0.86 ***	0.79 **	-0.19	0.29	-0.52	0.76 **	0.65 *	0.61 *	1	
BI[§]	0.07	0.48	-0.17	-0.06	-0.32	0.41	-0.28	-0.24	-0.01	-0.63 *	1

† Watershed drainage area
†† Channel slope
# Total Impervious area for upslope watershed
Watershed percent forest cover
§ Macroinvertebrate indices described in text
* $P < 0.05$
** $P < 0.01$
*** $P < 0.001$

Table 5. Significant variables in stepwise regression analysis of RGA and EPT Richness for high-gradient reaches at three different scales for TIA.

	RGA (N = 17)		EPT Richness (N = 14)	
Predictor Variables	Coefficient#	P	Coefficient#	P
Upslope Area†	Model R^{2*} = 0.80		Model R^{2*} = 0.90	
TIA	-0.151	< 0.001	-0.634	< 0.001
% Cobble	N.S.	N.S.	0.017	0.004
Local Area†	Model R^{2*} = 0.66		Model R^{2*} = 0.77	
TIA	-0.113	< 0.001	-0.420	0.007
Drainage Area§	N.S.	N.S.	0.173	0.080
% Sand	N.S.	N.S.	-0.014	0.071
Corridor†	Model R^{2*} = 0.52		Model R^{2*} = 0.55	
TIA	-0.050	0.005	N.S.	N.S.
Drainage Area§	0.049	0.099	0.282	0.034
% Sand	N.S.	N.S.	-0.023	0.026

* Model Adjusted R^2.

† Scale at which percent TIA was measured (see Figure 2).

Regression equation coefficient indicating the direction the variable relates to RGA and EPT Richness

§ Watershed drainage area for all models.

N.S. = Not significant in stepwise regression analysis and not included in model.

Table 6. Summary of significant ANCOVA results for ICM analyses and DHG regressions for high-gradient reaches. No significant interactions found for EPT models.

DHG Regressions

Response Variable	Model	Predictor Variables WT	DA	WT*DA
Width*	WT, DA, WT*DA	**0.001**	**< 0.001**	**0.001**
Depth*	WT, DA, WT*DA	0.800	**< 0.001**	0.453
Area*	WT, DA, WT*DA	**0.043**	**< 0.001**	0.079

ICM Analyses

Response Variable	Model	WT	Predictor Variables S	DA	WT*S	WT*DA
RGA	WT, S, WT*S	0.613	0.580	---	**0.011**	---
	WT, DA, WT*DA	**< 0.001**	---	0.490	---	**0.037**

Significant ($\alpha = 0.05$) variables and interactions shown in bold
WT = Watershed type; S = Channel slope; DA = Drainage area
* Bankfull dimensions
--- Not applicable

Figure 1. Site map of study watersheds in northwestern Vermont.

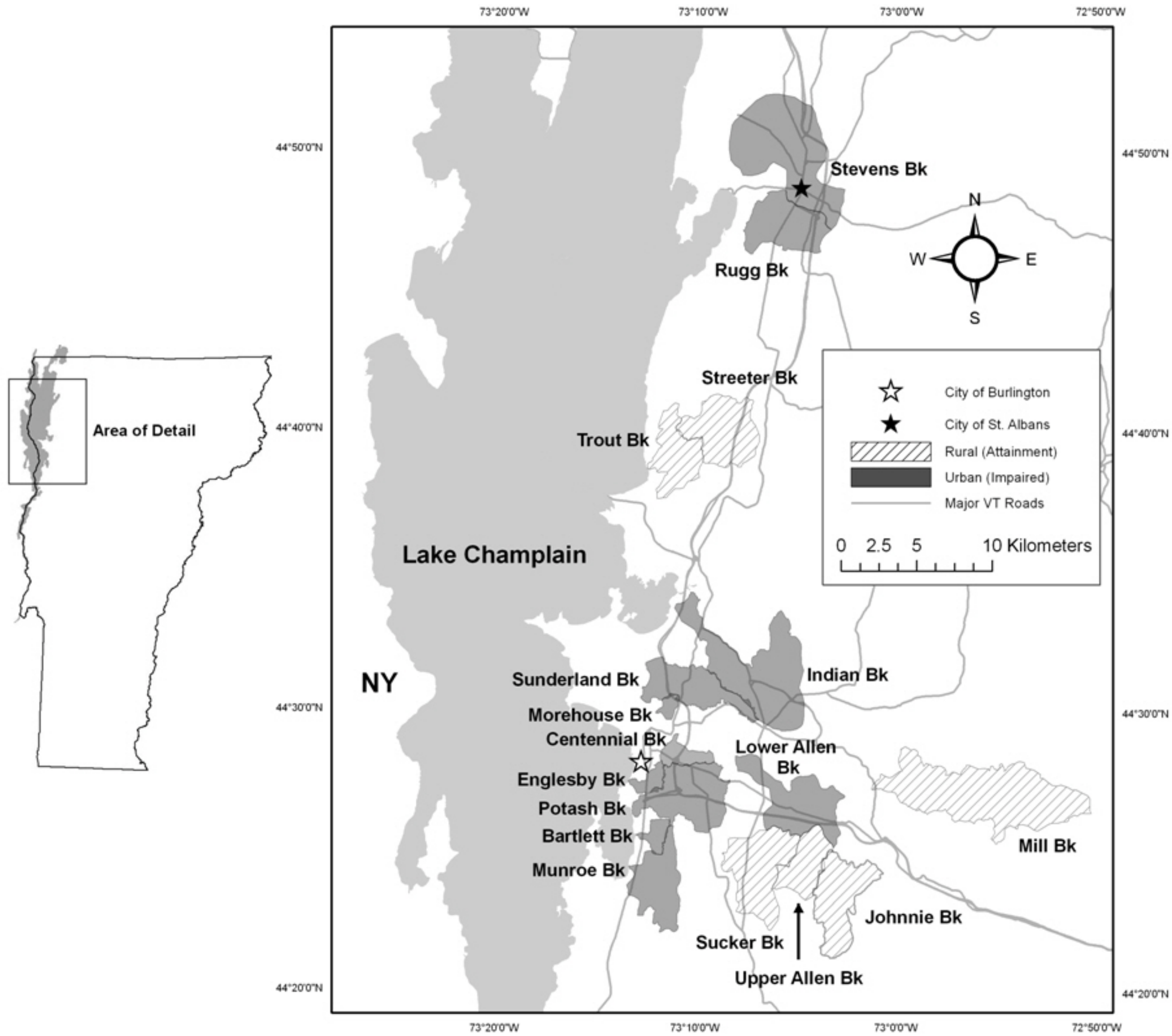


Figure 2. Conceptual map of spatial scales used to measure TIA

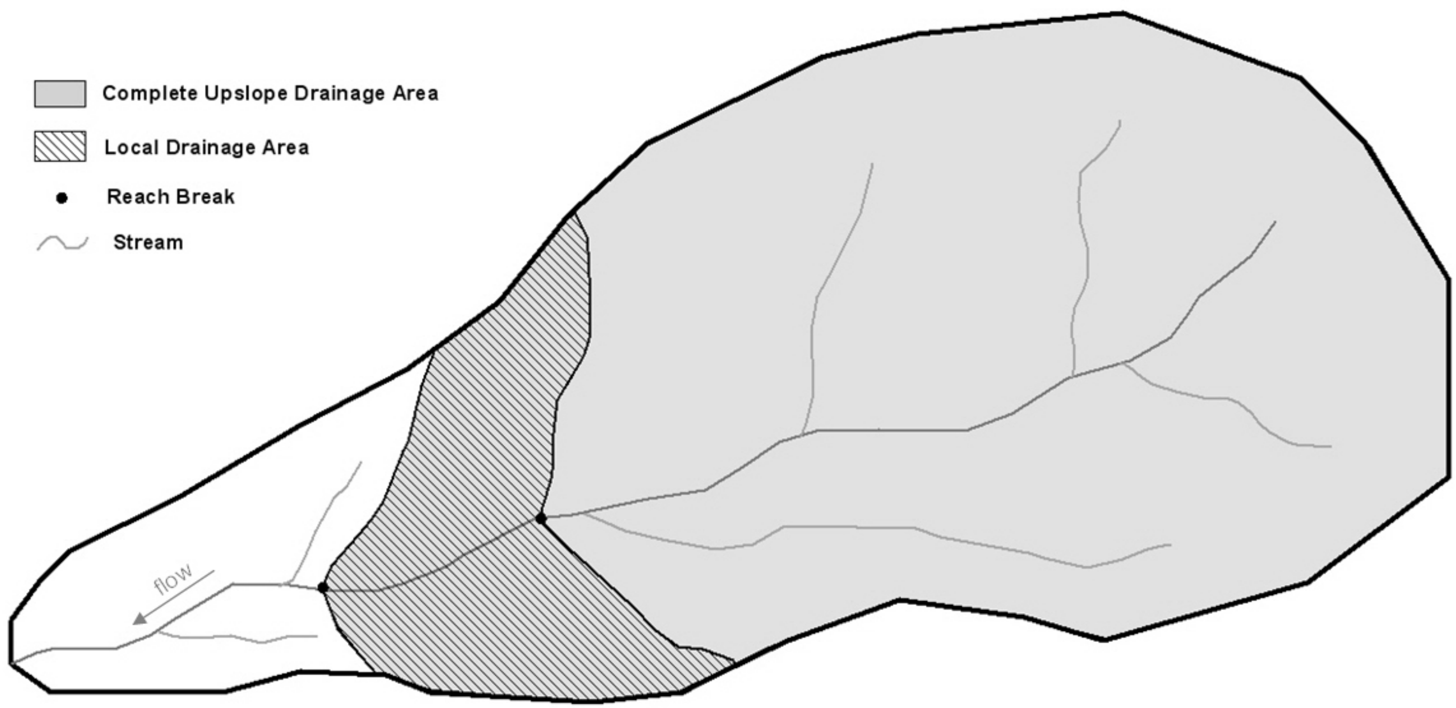


Figure 3. Plot of the relationship between RGA and percent upslope TIA for high-gradient study reaches. Categorical groupings of physical stream condition provided on right (VTDEC, 2005).

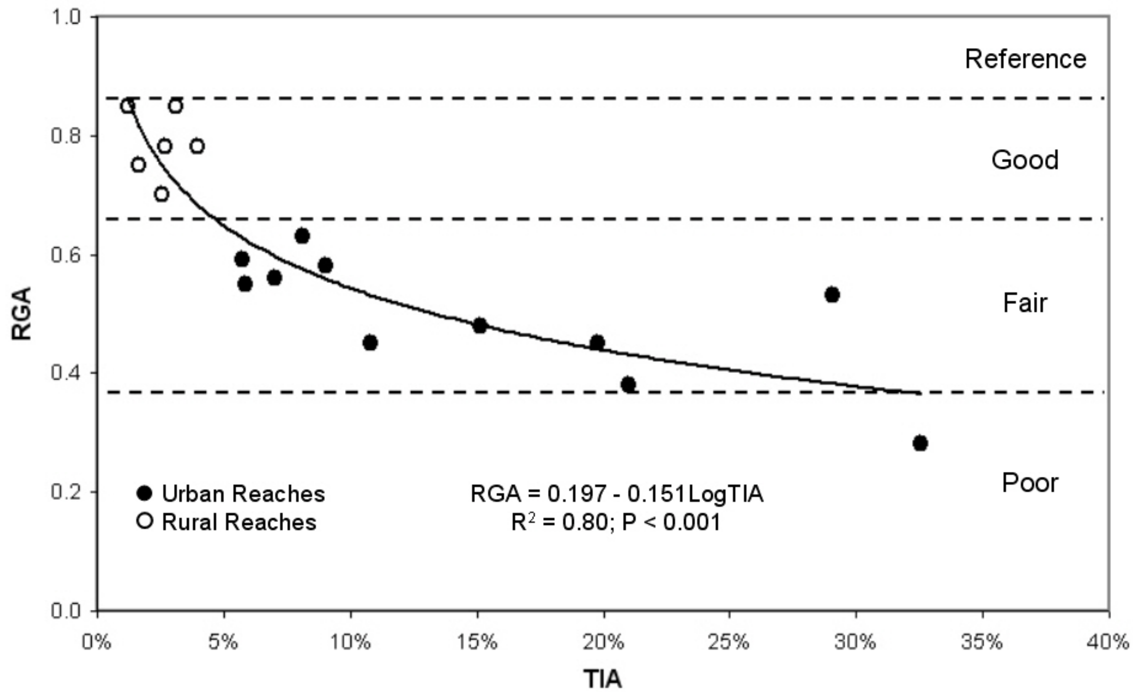


Figure 4. Plot of relationship between EPT richness and percent upslope TIA for high-gradient study reaches.

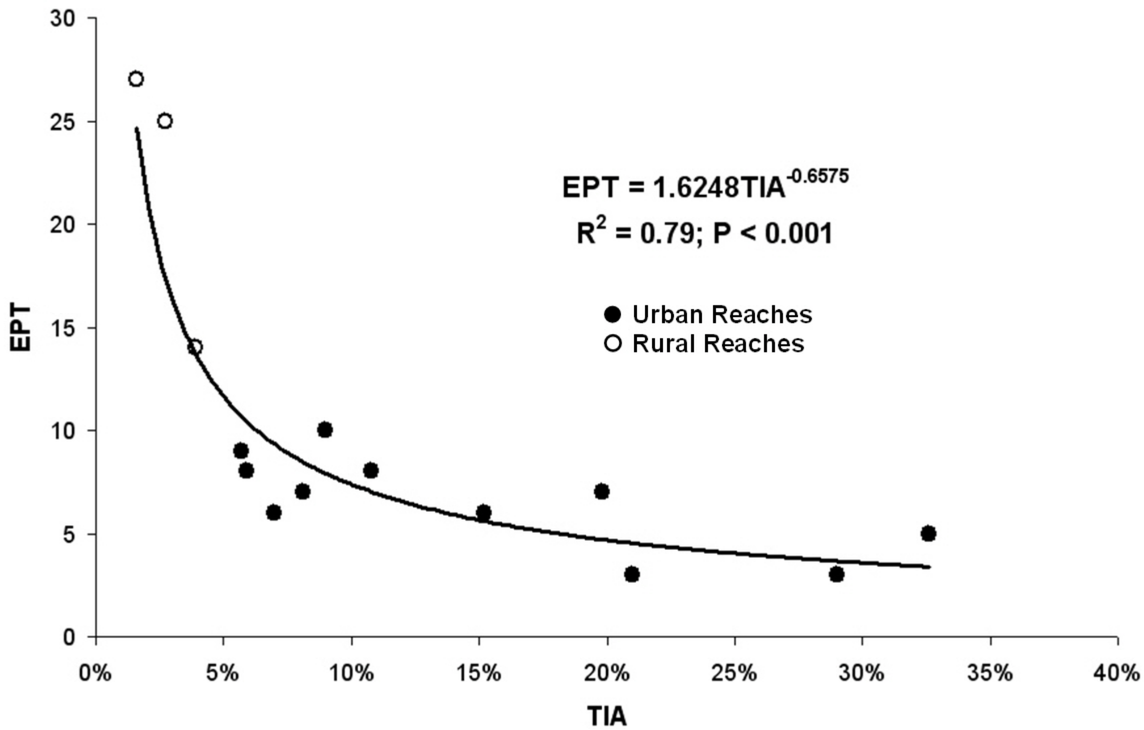


Figure 5. Plot of the relationship between RGA and percent upslope TIA for low-gradient study reaches. Categorical groupings of physical stream condition provided on right (VTDEC, 2005).

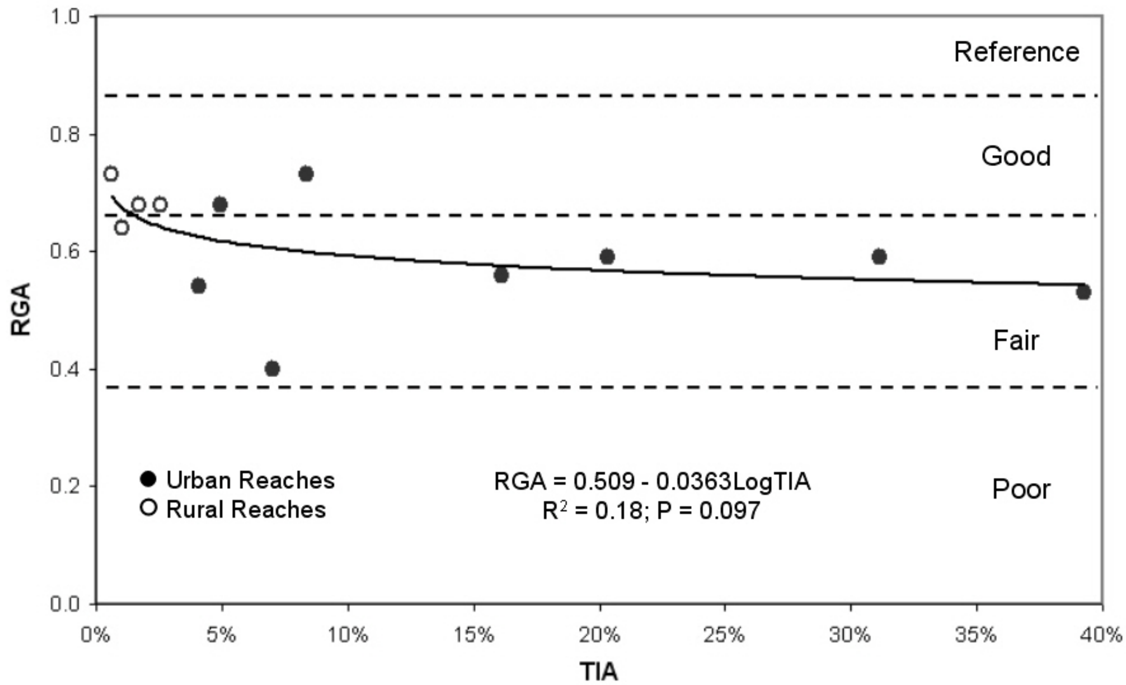


Figure 6. Plot of relationship between RGA and the interaction between watershed type and channel slope for high-gradient study reaches.

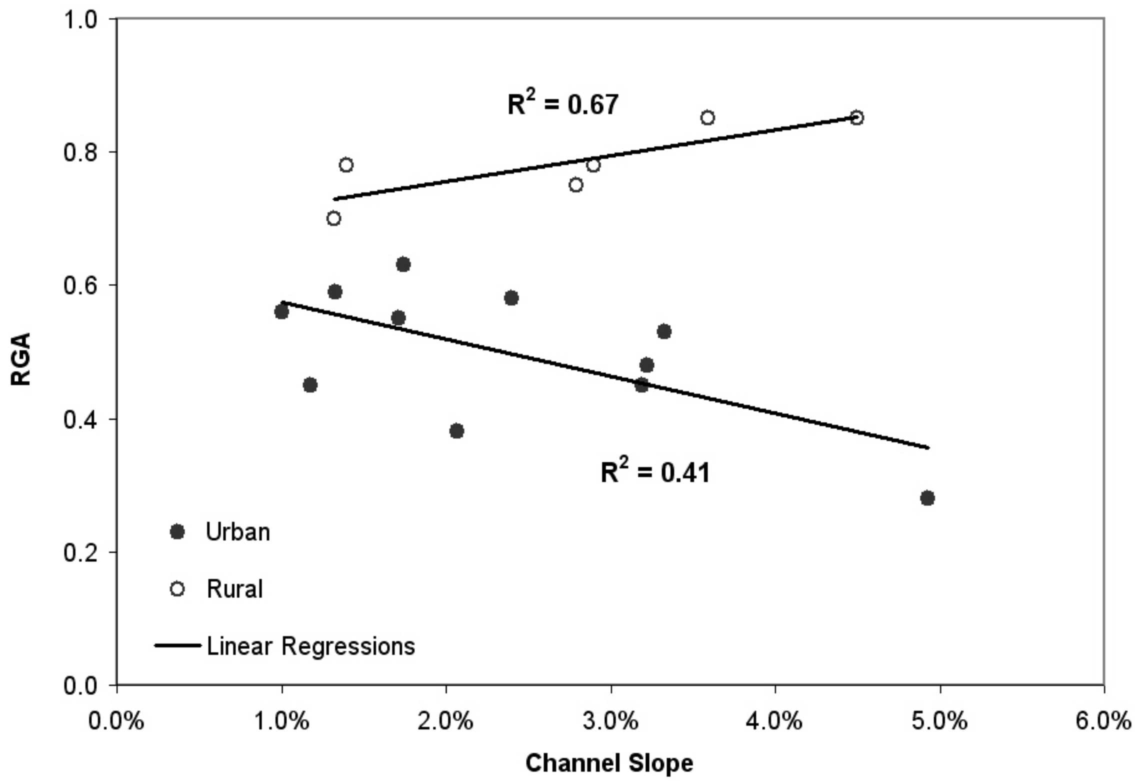


Figure 7. Plots of DHG regressions for bankfull channel width and depth for urban and rural high-gradient stream types. Regression equations noted on plot.

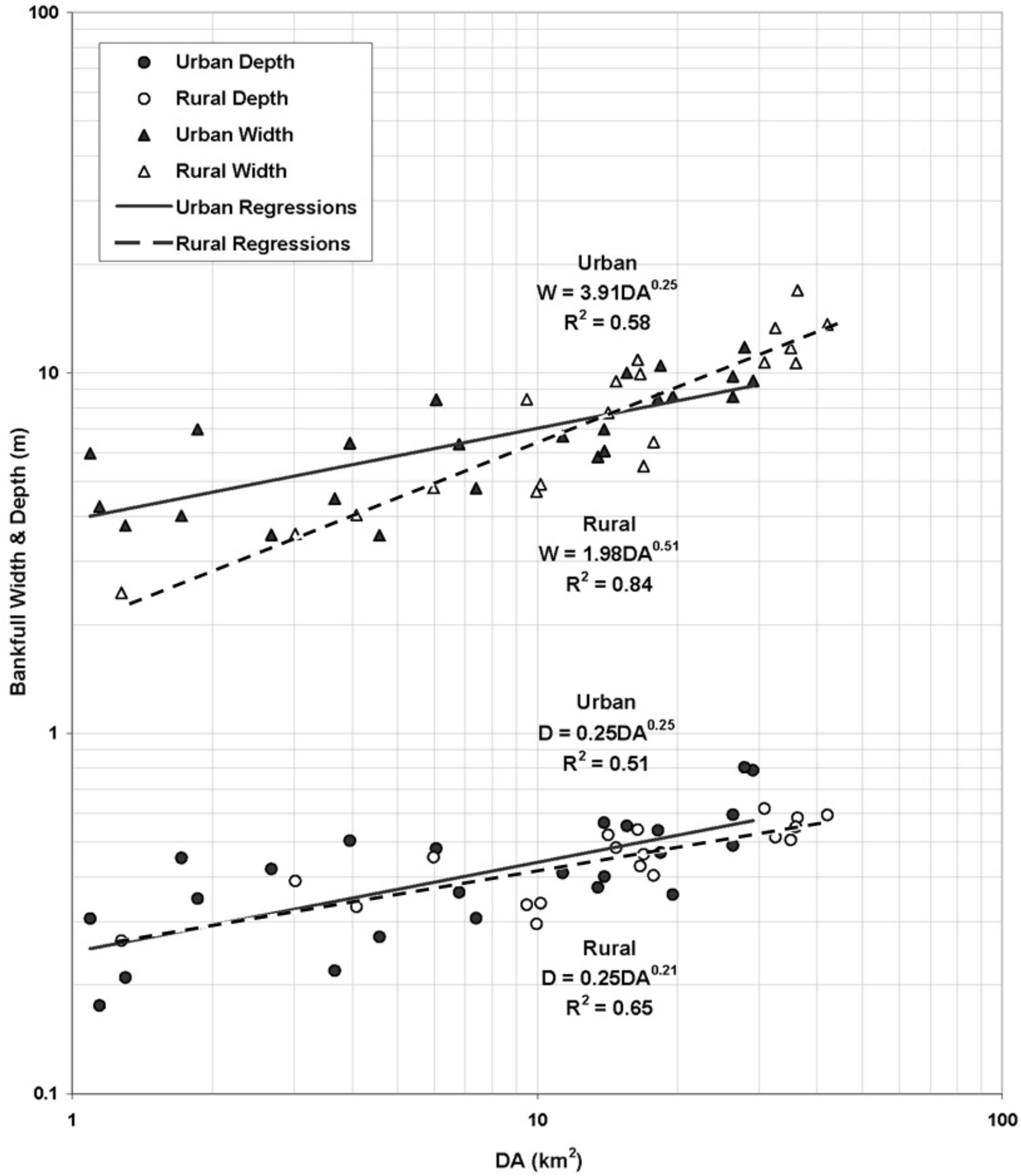
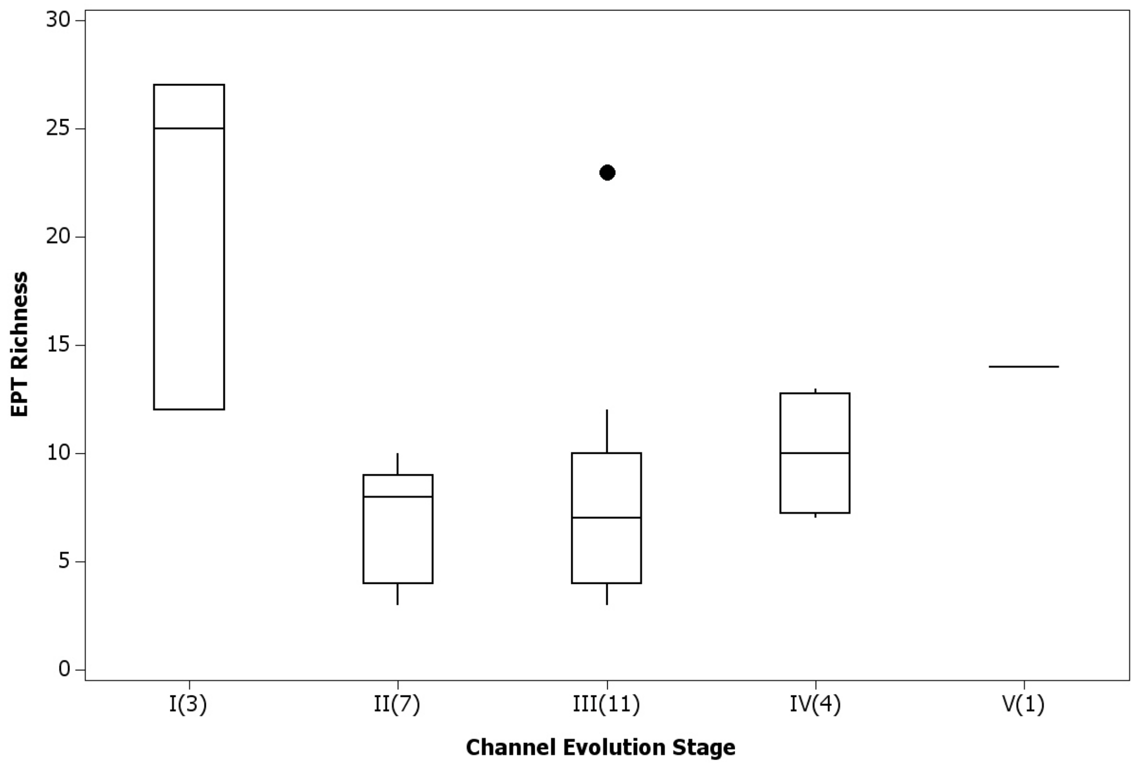


Figure 8. Boxplots of EPT richness for channel evolution stages for high-gradient stream reaches (N = 26). Number in parentheses is macroinvertebrate sample size for channel evolution stage. Boxes represent first and third quartiles (Q1 and Q3), with median value shown as the line between. Whiskers represent range of data within limits 1.5 (Q1 – Q3), with outliers represented by a dot.



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CHAPTER 3

RESPONSE OF CHANNEL WIDTH TO URBANIZATION: A TRANS-REGIONAL SCALE-DEPENDENT TREND

Abstract

Downstream hydraulic geometry (DHG) regressions are commonly used to model changes in channel form along the channel network caused by increasing discharge. DHG regressions are being promoted as tools for stream restoration projects in urban and rural watersheds where natural channel dimensions are sought as a basis for engineering designs, but their limitations in the urban environment have been unexplored to date. In this analysis, we tested the hypothesis that the channel widening response to urbanization in small watersheds is scale-dependent across physiographic regions. We analyzed urban and rural DHG regression parameters published for different physiographic regions of the United States and found a consistent response of channel width to urbanization that is dependent on the scale of the watershed area up to areas of ~50 km^2, a scale that includes first- to third-order headwater streams that are often directly impacted by urban and suburban development. This trans-regional response was observed across different environmental conditions such as climate, lithology, and stream bank vegetation. The results suggest that lower-order headwaters channels may be more susceptible to geomorphic adjustment than higher-order channels. This an important consideration for land use planners in urbanizing watersheds and stream restoration professionals intending to use DHG regressions for channel restoration designs.

Introduction

The number of river restoration projects across the United States has increased exponentially since the 1980's and projects associated with urban impacts rank among the most costly of all restoration types (Bernhardt et al., 2005; Bernhardt and Palmer, 2007). Despite high costs, increased urbanization in small watersheds across the United States has forced local and regional planners to address stream channel restoration needs (Riley, 1998). The impacts of urbanization on hydrology and geomorphology have been studied for many years (Leopold, 1968; Wolman, 1968; Hammer, 1975), however stream channel response been shown to differ across physiographic regions of the United States (Paul and Meyer, 2001) and the world (Chin, 2006). The development of models describing the response of stream channels to urban impacts, including conceptual models of channel evolution (Schumm et al., 1984) and quantitative models of hydraulic geometry (Doll et al., 2002), will contribute to our understanding of this response across different physiographic regions. Additionally, further exploration of hydraulic geometry regressions, which are a pillar of the widely used natural channel design approach (Rosgen, 1996), will determine their utility and limitations in urban stream channel restoration efforts.

Hydraulic geometry regressions were first developed by Leopold and Maddock (1953) using discharge to quantify and predict changes in channel dimensions, termed downstream hydraulic geometry (DHG). In self-formed alluvial channels, the DHG relationship for bankfull channel width is predicted using the following power equation:

$$W = \alpha Q^{\beta} \qquad (1)$$

where W is the bankfull channel width (m), Q is bankfull discharge (m^3/s), α is the regression coefficient, and β the regression exponent (and the slope of regression line). Numerous studies have shown the predictive utility of DHG regressions across diverse physiographic regions (Dunne and Leopold, 1978; Rosgen, 1994) and their limitations across different spatial scales, substrates and vegetation types (Wohl, 2004; Anderson et al., 2004). The values for exponent β in DHG regressions developed from different environments and regions of the world are remarkably constant, rarely deviating significantly from 0.5 in non-urban watersheds (Soar, 2000). When drainage area is used as a surrogate for Q, β is found to be consistently near 0.45 in non-urban watersheds (Castro and Jackson, 2001; Bent, 2006; Fitzgerald, 2007).

Since the 1970's, researchers have quantitatively examined the effect of urbanization on channel form using DHG regressions (Fox, 1976; Morisawa and LaFlure, 1979). Fox (1976) compared urban and rural DHG regressions in the Maryland Piedmont using Q as the independent variable. The resulting DHG regression parameters reported by Fox show a large amount of variability in channel geometry response across different geomorphic features (e.g., riffle), with no significant difference between the urban and rural regression equations. In contrast, Morisawa and LaFlure (1979) found that β tended to be smaller in more urbanized watersheds and closer to 0.5 in rural watersheds. More recently, Hession et al. (2003) reported that the influence of urbanization on channel width in gravel bed rivers of the Piedmont region was not significant without accounting for variations due to drainage area. Fitzgerald (2007) also noted a scale-dependent widening response to urbanization in gravel and cobble bed channels in Vermont.

Urban DHG regressions have been unexplored across physiographic regions to date, and little is known about their applicability to urban stream channel restoration. The purpose of this analysis was to test the hypothesis that the channel widening response to urbanization is scale-dependent, and to further hypothesize as to the mechanics of this response. In this study, parameters from urban and rural DHG regressions developed in different physiographic regions of the United States were used to explore this response. DHG width equation parameters are analyzed for: (1) seven urban and rural DHG regressions pairs found in published literature, and (2) regional rural DGH regressions readily available through the National Water Management Center (NWMC).

Methods

Urban-Rural Data Set

We reviewed published literature for DHG regressions comparing urban and rural watersheds across the United States. DHG regressions reviewed in this analysis reported bankfull channel width, depth and cross-sectional area as dependent variables. Seven urban-rural DHG regression pairs comparing bankfull channel width were reviewed and summarized (Table 1). In this study, regressions pairs are defined as one urban and one rural regression from the same regional environment. All DHG regressions use watershed drainage area (DA) rather than Q as the independent variable, as follows:

$$W = \alpha DA^{\beta} \qquad (2)$$

With the exception of one study (Doll et al., 2002), all urban-rural DHG regression pairs describe watersheds less than 50 km^2 in drainage area. The mean channel slopes reported across the urban-rural study areas ranged from 0.7% to 1.5%, and gravel was reported as the dominant bed substrate for each study (Table 1). Wohl (2004) has suggested that DHG regressions for a particular region that have coefficients of determination (r^2) of at least 0.5 describe well-developed DHG for that region. For the seven urban-rural regression pairs used in this analysis, 11 of 14 coefficients of determination for channel width were above 0.5 (Table 1) and the average value for all urban-rural pairs was never less than 0.5. Therefore, all parameter values from each of the 14 regression equations were used in the analysis.

NWCM Regional Rural Data Set

A separate data set of DHG regressions describing larger, rural watersheds was sought as a reference for comparison to the rural regressions of the urban-rural data set. While dozens of DHG regressions for rural areas across the world have been published since the 1950's, we chose a readily available group of publications found on the NWMC website. The NWMC is a center within the Natural Resource Conservation Service and has developed a website to organize DHG regressions by U.S. physiographic provinces. At the time of our analysis, eight independent DHG regressions published by Federal and State agencies were available on the NWMC website (Table 2). All DHG regressions selected for this data set use watershed drainage area as the independent variable

(Equation 2) and span a wide range of drainage areas (1.0 to 1170 km^2). Although the inclusion of additional DHG regressions from other sources could have increased the sample size of this dataset, the advantages of exclusively using this data set include: (1) consistent methodology for the development of DHG regressions; (2) the data are readily available and organized for a wide range of physiographic provinces.

Statistical Analyses

Although the equation parameters followed normal distributions (Anderson-Darling test with $\alpha = 0.05$), a conservative approach using a non-parametric Mann-Whitney test of the population medians was taken due to the small sample size of the data sets. Tests were performed to compare the populations of both regression parameters (α and β) for the following datasets: (1) urban versus rural (paired equations), (2) urban versus NWMC regional rural, (3) rural versus NWMC regional rural.

Results

There are significant differences between the paired urban and rural DHG parameter populations (Table 1). Boxplots depict the variability of α and β parameter populations for urban and rural equations (Figure 1). The urban α parameter population is significantly higher than the rural α parameter population ($p = 0.005$) and the urban β parameter population is significantly lower ($p = 0.004$) than the rural β parameter population (Table 1). No significant differences between the rural and NWMC rural datasets were noted in the analysis of either parameter population ($\alpha = 0.05$), indicating that the rural regression parameters from the small watersheds of the urban-rural data set were not anomalous.

The lower β parameter values and higher α parameter values observed in urban DHG equations reflect a consistent response of the mean channel width to watershed urbanization. Figure 2 depicts how this response has been observed on average across drainage area scales for the urban and rural watersheds. For six of the seven studies, urban stream channels are on average wider than their rural counterparts at smaller drainage areas, with this effect diminishing between drainage areas of approximately 10 to 15 km^2. The only urban-rural study that did not produce DHG regressions which intersect (widening effect exists at all scales) is the study containing data from the greatest range of drainage areas (Doll et al., 2002).

Discussion

Hession et al. (2003) noted the scale-dependence of the channel widening response of streams to urbanization in the small watersheds they studied in the Piedmont region of the eastern U.S. Our study extends this observation and presents the first trans-regional summary analysis of urban and rural DHG regressions. That a similar response has been observed across small watersheds with contrasting lithologies (Allen and Narramore, 1985) and vegetation types (Hession et al., 2003) supports the hypothesis that a similar trend would be observed across many regions of the country, and perhaps the world.

In exploring why this scale-dependent response was observed in seven of eight urban-rural studies, we hypothesized that two effects could contribute to scale-dependence: (1) the degree of urbanization is higher in smaller drainage areas, thereby causing a disproportionately greater alteration of the hydrologic regime in headwaters channels; (2) there are significant differences in channel slope between urban and rural study sites. Analysis of available data indicated that only one study (Brown, 1999) had a significant negative correlation between drainage area and percent watershed urbanization. With respect to slope, none of the studies from the urban-rural data set showed significant differences between the median channel slopes of the urban and rural study sites ($\alpha = 0.05$). Thus, neither the degree of urbanization nor slope seem to explain the observed differences in DHG in urban and rural streams.

Additional and more detailed data sets are needed to test whether other natural or anthropogenic landscape variables can explain the scale-dependent widening response in urban watersheds. It is possible that an interaction between the altered hydrologic regime and other inherent geomorphic characteristics which vary from headwaters to response zones, such as floodplain access, bed substrate, and channel sinuosity (Leopold et al., 1964), could help explain the response. Channel evolution stages (Schumm et. al, 1984), which describe the temporal response of channel geometry and planform to stressors, may also progress at different rates due to a gradient in floodplain and channel geometry across watershed scales. In small watersheds in the Pacific Northwest, Booth (1990) noted that lower-order, higher-gradient urban stream reaches are most susceptible to rapid channel incision processes, and that high sheer stresses in low-gradient, higher-order streams produced only minimal bed lowering. Channel incision processes are typically followed by bank failure and channel widening, and lower-order channels may be progressing more rapidly through channel evolution stages than higher-order channels. Further investigation of channel evolution processes at different scales in the urban environment is needed to understand adjustments along the channel network.

The effect of large woody debris (LWD) on channel widening may also be scale-dependent in small urban watersheds, and may interact with the hydrologic regime changes brought on by urbanization. Anderson et al. (2004) reviewed studies documenting the effects of riparian vegetation on channel form, and concluded that streams with woody bank vegetation are wider than those with herbaceous vegetation in watersheds with drainage areas less than 10 km^2. Anderson et al. (2004) suggest this response may be explained by a combination of higher LWD loading rates (Diez et al., 2001) and lower long-term LWD removal rates (Keller and Swanson, 1979) observed in headwaters reaches, and the increase in local erosion rates associated with these conditions (Trimble, 1997; Davies-Colley, 1997). Greater LWD loading and retention in headwaters reaches may exacerbate the channel erosion processes caused by increased runoff from impervious surfaces, and this effect may decrease in a downstream direction with reduced LWD loading (Diez et al., 2001).

Discharge scaling in urbanized watersheds may also be an important area of research which deserves further attention. Hydrologists have long documented the shape of storm hydrographs at different scales within rural watersheds. Dunne and Leopold (1978) described a dampening of storm hydrograph limbs with increases in drainage area, due to greater attenuation of peak flow volumes as floodplain access increases in a downstream direction. Responses in channel form are likely to be less affected by changes in the rise and fall storm hydrographs, and more so by changes in the duration of time that geomorphically significant stormflows are exceeded (Leopold et al., 1964). However, Glaster et al. (2006) reported a non-linearity in the scaling of peak flow discharge in an urban watershed, suggesting a scale-dependence of effective water loading. Disproportionate hydrologic loading due to uneven spatial distributions of impervious cover in urbanizing watersheds may also contribute to a scale-dependent response in channel form.

The results of this analysis have important implications for the practices of urban stream channel restoration. Since Rosgen (1996) suggested the use of DHG regressions as tools for natural stream channel design, numerous Federal and State agencies have adopted this approach. In many urban stream channel restoration projects, limitations of stream discharge data preclude the development of DHG regressions using Q as the independent variable. In these situations, DHG regressions developed using drainage area as the independent variable could be misused if the issue of scale-dependence is ignored. Failure to recognize scale-dependent differences in the response of channel geometry to urbanization could lead to improperly scaled channel restoration designs and project failure. Regional stream restoration projects in small urbanizing watersheds would benefit from additional field efforts to document urban-rural DHG comparisons before significant resources are spent reconfiguring channel geometry.

Acknowledgements

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Table 1. Characteristics of Urban-Rural Datasets

Author	Geographic Area	Drainage area range (km^2)	Channel Slope*	Bed Substrate†	α Urban§	α Rural§	β Urban§	β Rural§	r^2 Urban§	r^2 Rural§
Allen and Narramore (1985)	Shale Lithology, North Texas	1.3 - 26	0.4 - 0.8%	gravel	6.22	3.52	0.25	0.43	0.33	0.72
Allen and Narramore (1985)	Chalk Lithology, North Texas	1.3 - 26	0.4 - 0.8%	gravel	7.17	2.38	0.30	0.67	0.66	0.83
Brown (1999)	Piedmont, Maryland	0.6 - 26	0.7%	gravel	4.52	2.20	0.22	0.48	0.52	0.56
Doll et al. (2002)	Piedmont, North Carolina	0.5 - 332	0.7%	coarse sand;gravel	5.43	3.14	0.33	0.36	0.88	0.91
Fitzgerald (2007)	Lake Champlain Basin, Vermont	1.1 - 42	1.5%	gravel;cobble	3.91	1.98	0.25	0.51	0.58	0.84
Hession et al. (2003)	Piedmont: PA, NJ, DE, MD (Forested)	0.4 - 51	0.9%	gravel	5.83	4.15	0.13	0.30	0.45	0.82
Hession et al. (2003)	Piedmont: PA, NJ, DE, MD (Non-Forested)	0.4 - 51	0.9%	gravel	3.86	1.97	0.12	0.46	0.24	0.91
				Statistics						
				median value	5.43	2.38	0.25	0.46	0.52	0.83
				p value††	0.005		0.004		N.A.	

* Mean channel slope for all reaches in dataset (if available), otherwise range of channel slopes

† Median channel bed substrates observed in dataset for coarse sand (0.5-2 mm), gravel (2-64 mm) and cobble (64-256 mm)

§ Parameters and coefficients of determination from DHG regression equation

N.A. = Not Applicable

†† Probability that parameters from urban and rural regressions are from the same population.

PA - Pennsylvania; NJ - New Jersey; DE - Delaware; MD - Maryland

Table 2. Characteristics of NWMC Rural Datasets

Author	Geographic Area	Drainage area range (km^2)	$\alpha^{§}$	$\beta^{§}$	$r^{2§}$
Chaplin, J.J. (2005)	Non-carbonate Lithology: PE, MD	8.9 - 554	2.91	0.45	0.81
Dudley, R. W. (2004)	Coastal & Central Maine	7.5 - 775	1.42	0.52	0.82
Keaton et al. (2005)	Mid-Atlantic Valley/Ridge: MD, VA, WV	0.3 - 642	2.50	0.44	0.92
Kuck, T.D. (2003)	Southwest Oregon	1.0 - 1185	2.35	0.42	0.94
Mulvihill et al. (2005)	New York Region 6	2.6 - 751	3.45	0.42	0.79
Sherwood and Huitger (2005)	Ohio Region A	1.3 - 1770	3.89	0.36	0.91
Vermont DEC (2006)	Vermont Statewide	7.8 - 361	2.57	0.44	0.91
Westergard et al. (2005)	New York Region 5	1.8 - 863	2.59	0.46	0.90
		median value	2.58	0.44	0.91

§ Parameters and coefficients of determination from DHG regression equation

N.A. = Not Applicable

PA - Pennsylvania; MD - Maryland; VA - Virginia; WV - West Virginia

Figure 1. Boxplots of α and β regression parameters for the urban-rural data set. Boxes represent first and third quartiles (Q1 and Q3), with median value shown as the line between. Whiskers represent range of data within limits 1.5 (Q1 – Q3).

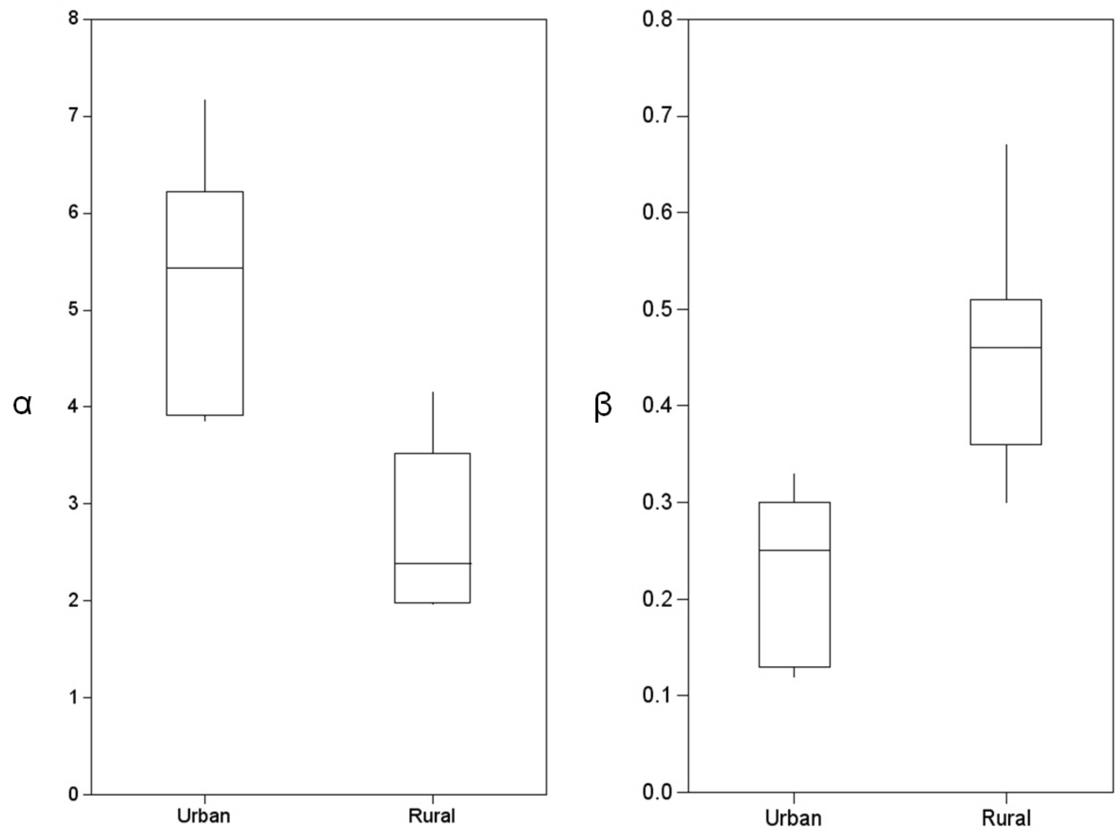
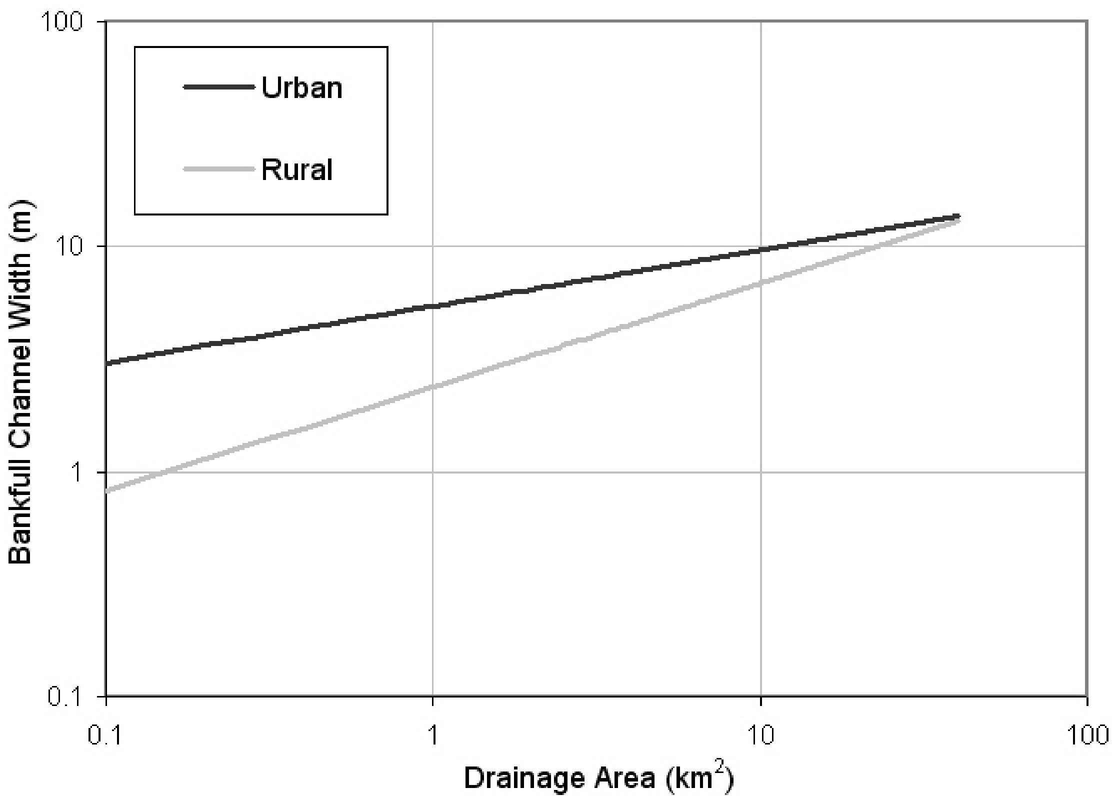


Figure 2. Response of channel width to urbanization for urban-rural dataset. Regression lines generated using median values of α and β parameters (Table 1) and the DHG power equation (equation 1). The widening effect is greatest in drainage areas less than 10 km^2.



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APPENDIX A: LOCATION OF WATERSHED REPORTS AND DATA

The UVM-VTANR data collection effort in summer 2005 resulted in reach-scale SGA data for a total 145 reaches from 14 watersheds (Table 1). An additional two watersheds in vicinity of the City of St. Albans, Rugg and Stevens Brooks, were assessed by the Lake Champlain Committee. All data collected for these 16 watersheds, including the data used for the analyses in Article 1 of this thesis, is publicly accessible on the VTANR Database Management System (DMS) using the following link:
https://anrnode.anr.state.vt.us/ssl/sga/security/frmLogin.cfm

In addition, reports which summarize the watershed and geomorphic conditions of the impaired watersheds (Table 1) have been completed. The reports contain the following elements: (1) narrative descriptions of the watershed zones and reaches assessed; (2) preliminary project identification information; (3) Quality Assurance/Quality Control (QA/QC) summaries of the data; (4) stream cross-sectional plots; (5) reach summary statistics; (6) subwatershed and reach maps. These reports will be made available on the VTANR River Management Program website at the following link:
http://www.anr.state.vt.us/dec/waterq/rivers.htm

Table 1. Summary of Watershed Reaches and Reporting

Watershed	*Water Quality Status*	*Number of Reaches Assessed*	*Final Report?*
Allen Bk	Impaired	14	Yes
Bartlett Bk	Impaired	6	Yes
Centennial Bk	Impaired	7	Yes
Englesby Bk	Impaired	6	Yes
Indian Bk	Impaired	23	Yes
Morehouse Bk	Impaired	5	Yes
Munroe Bk	Impaired	13	Yes
Potash Bk	Impaired	16	Yes
Sunderland Bk	Impaired	10	Yes
Johnnie Bk	Attainment	11	No
Mill Bk	Attainment	21	No
Streeter Bk	Attainment	1	No
Sucker Bk	Attainment	7	No
Trout Bk	Attainment	5	No
	Total Impaired Reaches:	100	
	Total Attainment Reaches:	45	
	Total:	145	

APPENDIX B: SPATIAL ANALYSIS OF TOTAL IMPERVIOUS AREA

Land use data derived from two separate sources for the study area was utilized to quantify Total Impervious Area (TIA) percentages for each drainage area. Statewide Landsat imagery collected in 2002 using a 30 m grid was processed by UVM's Spatial Analysis Laboratory (SAL), resulting in the following four spectral classes: (1) forest; (2) urban; (3) open (agricultural and open recreational uses); (4) water and other (SAL, 2005). In addition, a separate dataset of TIA derived from high-resolution Quickbird satellite scenes collected between 2003 and 2005 was utilized (Morrissey and Pelletier, 2006). The multispectral bands (2.4 m resolution) from the Quickbird scenes were analyzed by SAL using Definiens eCognition® software to classify the data into three classes: (1) impervious; (2) pervious; (3) water.

Quickbird-derived TIA data was only available for a select group of watersheds during the time of this analysis. Given this limitation, a correlation analysis was performed using the Landsat-derived urban class and the Quickbird-derived TIA class for 4 of the 16 study watersheds. The dataset used for the correlation analysis included 40 independent subwatersheds with a wide distribution of drainage areas (ranging from 0.07 km^2 to 3.8km^2) and TIA percentages (ranging from 1.2% to 40.6%). The analysis resulted in a robust linear relationship that was used to calculate TIA for all study watersheds at each spatial scale (Figure 1).

Figure 1. Plot of relationship between Urban Land Cover and TIA for 40 independent subwatershed areas from 4 of the study watersheds.