ABSTRACT

Title of Document: FISH MOVEMENT, HABITAT SELECTION, AND STREAM HABITAT COMPLEXITY IN SMALL URBAN STREAMS

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Urbanization impacts have become more evident in the last 30-50 years, due to human population increase and subsequent land use change. Many aspects of stream ecosystems are influenced including hydrology, geomorphology, water quality, ecosystem function, riparian vegetation, and stream biota. Effects of urbanization on ecosystem structure and function are discussed, and the urban stream syndrome is introduced in Chapter 1. Chapter 2 reports differences in stream fish assemblages in the eastern Piedmont and Coastal Plain of Maryland, USA due to urbanization, and establishes a foundation for hypotheses presented in subsequent chapters. Chapter 3 describes a physical habitat survey that attempts to understand what instream and channel habitat attributes change across the urban–rural gradient (0–81% urban land use; ULU). While changes in stream habitat appear at 30% ULU, significant impacts occurred once a watershed has >45% ULU, at which point stream channels can not accommodate the power and intensity of impervious surface runoff. Fish habitat patch selection is examined in Chapter 4, which involved instream habitat manipulation experiments. I tested fish selection response of instream habitat using three treatments (woody debris, shade, and both) in first order urban (>60% ULU), suburban (27-46% ULU), and rural (<15% ULU) eastern Piedmont streams in Maryland. Blacknose dace (BND) *Rhinichthys atratulus* and creek chub (CKB) *Semotilus atromaculatus* selected shade and woody debris combined significantly more than other treatments in rural and suburban streams. Urban fish selected the shade treatment the most of all enhancements. CKB who selected the enhancement were significantly larger than those found in the control. Urban fish prefer shaded habitat providing overhead protection due to the general lack of habitat complexity in urban channels. CKB behavior may indicate intraspecific competition, particularly between juvenile and adult individuals for prime habitat positions. Chapter 5 presents a fish movement study, comparing rural and urban fish population behaviors. Urban BND and CKB displayed significantly larger home ranges than rural fish. The rural fish movement distribution was more leptokurtic. Competitive interactions are suggested as the reason for greater movement in urban stream populations. Finally, conclusions are submitted with significant findings in Chapter 6.

FISH MOVEMENT, HABITAT SELECTION, AND STREAM HABITAT COMPLEXITY IN SMALL URBAN STREAMS

By

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Foreword

The research carried out and presented in Chapter 2 of this doctoral dissertation was previously published in 2005 in the Journal of the North American Benthological Society, volume 24, number 3, pages 643-655. I have received permission to reprint this article in my dissertation from the Journal of the North American Benthological Society. The results and conclusions of this piece served as a catalyst in developing my experimental hypotheses tested in subsequent research chapters. As a coauthor of this peer-reviewed journal article, I made significant contributions to the conceptual and analytical aspects of this jointly written study and therefore have included it in this doctoral dissertation.

Dedication

This doctoral dissertation is dedicated to Jeremy, my husband and best friend, who offered endless love and support during the completion of this work and gently pushed me to achieve higher goals. And to my son, Riley, whose tiny presence and life encouraged to meet my deadlines throughout the writing of this research, timing was perfect, and birth has made me complete.

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Chapter 1: An introduction to urban studies on stream ecosystems

Abstract

Urban stream ecosystems experience significant impacts due to upstream land use change within the watershed. Small streams provide important ecosystem services to downstream waters but are particularly susceptible due to their proximity to new development. An increase in the number of studies on urban streams has led to the conceptualization of the urban stream syndrome, which describes the substantial ecological and environmental degradation that occurs in these watersheds. Changes in hydrology, geomorphology, water quality, ecosystem function, riparian vegetation, and biotic communities are documented. Generally, urban streams exhibit a flashy hydrograph, altered geomorphology and channel stability, and decreased water quality, including increased temperature, sediment, and conductivity. A few studies of ecosystem function, particularly leaf breakdown, retention of organic matter, and nutrient processing, have shown reduced maintenance of ecosystem services. Riparian buffer function is also modified in urban stream ecosystems due to increased drainage connectivity. Finally, urban biotic communities display decreased species richness, increased tolerant species, and decreased sensitive species when compared to less-impacted stream communities. A variety of experimental approaches have been used to investigate urbanization impacts, including experimental manipulations, paired watershed design, and the use of land-use gradient to document these changes. Hypotheses and brief descriptions for the following research chapters which examine four studies of urbanization impacts are presented.

Urbanization impacts

The expansion and influence of urbanization on natural landscapes has been dramatic in the last half century and is becoming one of the most dynamic processes of global ecosystem change (Grimm et al. 2000). As anthropogenic impacts are integrated into aquatic, terrestrial, and atmospheric environments, ecosystems respond over highly complex spatial and temporal scales. Modifications of land cover and widespread land use change have immense consequences on aquatic ecosystems (Schlosser 1991, Allan 2004, Paul and Meyer 2001, Gergel et al. 2002, O'Neill et al. 1997). Small streams are particularly vulnerable to landcover changes and their associated effects due to their proximity to new development and the rate at which rural and forested land is converted into residential, municipal, and commercial uses (Feminella and Walsh 2005, Walsh et al. 2005b). Furthermore, issues of water quality and biotic integrity concern both human health as well as ecological structure and function. Management of water resources has already become and will become even more of a critical issue in the future. As the population size of the US increases to over 400 million by 2050 (projected; USCB 2000), the demand for water supply and food production will be compounded by land development consequences (Fitzhugh and Richter 2004). Although population increase is not consistently uniform across the country, a national trend of overall population growth is evident (Otterstrom 2003).

Nearly half of Maryland's streams are currently rated to be in poor condition, and urban development has had a pronounced impact on biotic integrity (Roth et al. 1999). In the last 30 years, the northern Piedmont region of Maryland has experienced an exponential growth in percent urban land cover (Griffith et al. 2003). At present rates of urban expansion in Maryland, the extent of urban land use (now ~16%) is predicted to grow to 21% in the next twenty years (Boward et al. 1999). As a result, impacts associated with this development may be expected to further degrade Maryland streams, as well as the Chesapeake Bay. At a time when significant attention is being devoted to restoring aquatic resources, impacts of continuing urbanization on stream health in Maryland need to be fully recognized.

The Chesapeake Bay watershed includes a large portion of Maryland, especially its most urbanized regions. The "State of the Bay" reflects the conditions of upstream tributaries as well as complex interactions that take place in areas localized around the Bay's shores. To reverse trends of ecological degradation occurring in the Bay today, remediation and further protection of its tributaries are essential. The renewed agreement, *Chesapeake 2000*, recognizes the need to preserve and protect every stream, creek, and river, promote stream corridor restoration, and develop sound land use practices (CBP 2000). Protection and restoration of streams are essential management practices that support better water quality and vitality of natural resources in the entire watershed, and the Bay itself.

In recent years, the number of studies that examine some aspect of urban stream ecosystems has increased considerably. Urban stream studies began to emerge in the early 1970's but did not gain the attention of many scientists until the late 1990's (Figure 1). Since then, there has been a steady increase in research dedicated to understanding the impacts of urbanization in stream networks across the world. As of March 2006, there were over 200 peer-reviewed documents on urban stream ecosystems (Figure 1). In 2003, the American Fisheries Society annual meeting and the Symposium on

Urbanization and Stream Ecology were held on the effects of urbanization on stream ecosystems (Feminella and Walsh 2005, Brown et al. 2005). Both of these symposia proceedings have granted a majority of support and attention to urban stream studies in recent years, and along with have contributed peer-reviewed publications to the stream literature base. In addition, the National Science Foundation funds research at two urban Long-term Ecological Research (LTER) sites, Baltimore Ecosystem Study and the Central Arizona – Phoenix LTERs. Research groups at these sites are working to quantify energy fluxes and spatial relationships within urbanized systems as well as to better understand how community behavior, socioeconomics, political structure, and land development affects the function of aquatic and terrestrial systems (BES 1998).

Over these last 35 years, a multitude of urbanization impacts on streams have been described. Recently, scientists summarized the major changes or symptoms that occur on a consistent basis in streams with heavily developed watersheds (Paul and Meyer 2001, Walsh et al. 2005b). The "urban stream syndrome" was proposed by Meyer et al. (2005) in their study of urban stream ecosystem function. Symptoms of the urban stream syndrome include a flashy hydrograph, elevated nutrient and contaminant concentrations, altered channel morphology and stability, and reduced species richness of stream biota (Figure 2, Walsh et al. 2005b). In comparison to rural streams, urban stream ecosystems exhibit decreased nutrient uptake with a concomitant increase in nutrient inputs, and increased stormflow discharge due to runoff from connected impervious surfaces (Figure 2). Riparian vegetation along the stream channel is modified, and its function is reduced as urban streams display wider channels. Increased water temperature, pool depth, and erosional scour are also indicated in urban stream systems

(Figure 2). Urban channel habitat complexity is reduced due to a lack of instream debris. In addition, there an increased number of tolerant fish species in urban streams (Figure 2). Although variability does occur across many ecosystems, there is a general consensus that these symptoms drive or lead to overall stream degradation in metropolitan areas. In the following section, I describe in detail the support as well as some of the controversy in scientific findings for the urban stream syndrome.

Hydrology

One of the most marked impacts of urbanization on stream networks is altered hydrology due to impervious surfaces and upstream land use (Arnold and Gibbons 1996, Paul and Meyer 2001, Groffman et al. 2003, Wheeler et al. 2005). Impervious surfaces are those regions of land that do not allow precipitation to enter the groundwater supply via infiltration through the soil column, such as parking lots, large building roofs, and roads (Arnold and Gibbons 1996). The impact of impervious surfaces has become increasingly prevalent in the study of stream habitat degradation (Moore and Palmer 2005, Jones et al. 1999, O'Neill et al. 1997, Roth et al. 1999, Wang et al. 2001, Walsh et al. 2001, 2005a). Generally, impervious surfaces are responsible for decreasing the capacity for infiltration, and increasing surface runoff, sheet erosion, sediment delivery, pollutants, and erosion and incision of stream channels due to drainage outfalls (Arnold and Gibbons 1996, Paul and Meyer 2001, Groffman et al. 2003). Precipitation that falls on impervious surfaces is directly routed to the stream channel, providing a dramatic increase in headwater stream discharge during and immediately after storms (Jones et al. 2000, Poff et al. 1997).

In less-impacted systems, water may flow longitudinally down a stream network, through lateral connections with soil water, or with vertical connections between the streambed and groundwater reserves. This connectivity within stream networks confers stable ecological function, defined by natural ranges of flow, storage, and transfer of energy and materials. Direct linkage between the stream channel and groundwater recharge produces more consistent, higher baseflow discharge. Conversely, if a stream is disconnected from adjacent land margins, there is greater risk of headwater streams drying up during the summer, altering the structure and function of stream systems (Groffman et al. 2003, Poff et al. 1997).

Modifications in water delivery through storm drains and sewers in highly urbanized regions can artificially increase the extent to which the surface of the watershed is connected to the stream network. Although connectivity is commonly used in landscape ecology models of patch dynamics (Wiens 2002), the drainage connection between impervious surfaces and the stream channel has recently been used as an indicator of urbanization effects (Arnold and Gibbons 1996, Hatt et al. 2004, Walsh et al. 2005a). Walsh et al. (2001) refer to this as "effective imperviousness" while Wang et al. (2001) have termed it "connected imperviousness". Connected impervious surfaces are those that are directly linked to stream channels via road drains, pipes, and underground channels. This connection generates a frequent disturbance regime, altering overall stream integrity, i.e. the physical, chemical, and biological features of the stream ecosystem, through complex pathways (Wheeler et al. 2005). As a consequence of the altered flow regime in urban stream networks, Konrad and Booth (2005) identified three principal hydrologic changes in urban streams. Compared to rural streams, urban streams experienced increased high-flow frequency, a relocation of water to storm flow from base flow, and increased daily variation in streamflow. Wissmar et al. (2004) and Roy et al. (2005b) demonstrated changes in stormflow magnitude in their studies. Urban baseflow may be lower than in forested watersheds (Klein 1979), yet some studies have found that this is not always the case (Konrad and Booth 2005, Brandes et al. 2005, Roy et al. 2005b). Baseflow discharge may not be lower necessarily, but if the channel has experienced widening due to erosion, channel depth may not be sufficient to support biota (Konrad and Booth 2005). Other causes for higher than expected baseflow may be due to contribution by leaky sewage or public water supply pipes (Paul and Meyer 2001).

Geomorphology

Changes in stream channel characteristics are also evident in urbanized watersheds due to altered flow regimes. Stream channels become unstable due to increased intensity of stormflow producing lateral and vertical scour (Groffman et al. 2003). These processes result in wider, incised streambeds (Hammer 1972, Trimble 1997, Bledsoe and Watson 2001, Hession et al. 2003, Roy et al. 2005a). In particular, Hammer (1972) found that streams adjacent to land with houses and sewered streets constructed more than four years prior exhibited significant channel enlargement. However, land developed less than four years and after 30 years ago did not display major changes in channel width (Hammer 1972). In newly developed watersheds where increased sediment loads are transported downstream, channel depth may decrease throughout the stream network (Clark and Wilcock 2000). However, some geomorphic studies claim that changes in potential stream power and thus channel stability are watershed-specific, and generalizations about urbanization cannot be made (Bledsoe and Watson 2001, Doyle et al. 2000).

The degree to which impervious surfaces are connected to the stream channel determines how severely stream channel morphology is degraded. Channelization and the extent to which a reach is piped drastically alter stream habitat channel structure (Paul and Meyer 2001). McBride and Booth (2005) argue that the extent of grassy land cover within the subwatershed and within 500 m of the stream channel in combination with the proximity of a road crossing best explains the physical condition of the stream channel. In this case, road and semi-impervious surfaces, like grassy land cover, present higher connectivity with the stream channel than other types of land cover. On the other hand, the ability of grassy riparian areas to trap and accumulate sediment was not reduced by urban stormflows in streams studied by Hession et al. (2003).

Most mature urban streams are devoid of fine sediment, as a result of years of sediment transport downstream (Groffman et al. 2003). However, in newly urbanizing watersheds, this is not always the case. Channel erosion is a primary source of sediment (Trimble 1997). As stated earlier, channel depth may decrease downstream due to accretion of transported sediment from upstream land use change (Clark and Wilcock 2000). Walters et al. (2003) examined stream morphology and water quality in relation to fish assemblages and found that urban stream water was more turbid, and channels were lined with fine sediment beds. However, slope of the stream channel predicted the dominant sediment size-class in this study. Thus, the morphological changes that occur in urbanizing and stable urban stream channels differ, and must be interpreted cautiously. Some differences may be due strictly to topography, soil composition, and climate.

Water quality

The earliest studies of urbanization effects on stream condition were related to the changes observed in water quality (Bryan 1971, Hordon 1973, Klein 1979). Prior to the 1970's, when the Water Quality Improvement Act of 1970 and Federal Water Pollution Control Act Amendments of 1972 (later amendments known as the Clean Water Act of 1977) were instated, untreated sewage, oil, and industrial effluents were discharged directly into river systems (Klein 1979). Many countries, including Brazil, still discharge untreated sewage into stream networks (Pompeu et al. 2005). However, contaminants still enter streams in this country as non-point source pollution degrading water quality due to state-state differences in discharge permits. Parameters frequently used to describe water quality include temperature, pH, dissolved oxygen, suspended sediment, conductivity, chemical pollutants and recently, concentrations of pharmaceuticals and personal care products.

Temperature of stream water is critical to the life history of many organisms, as well as stream processes. Reduction of riparian canopy providing shade in urbanized watersheds is a major source of increased stream temperature (Brasher 2003, Klein 1979, LeBlanc et al. 1997). Although Paul and Meyer (2001) claim that few studies actually document increased stream temperature in urban watersheds, increasingly more studies show this trend. Ambient temperature regimes around cities are many times referred to as having a "heat island effect", where stored heat from solar radiation is released from buildings and streets, often occurring at night (Kalnay and Cai 2003). Thus, the range of ambient air temperatures is shifted upwards, and may have a direct impact on diurnal temperature patterns in stream water as well. One study indicated that urban streams have higher summertime temperatures and lower winter temperatures than forested streams, with stormflows during summer reaching 10-15ºC higher than forested reaches as a result of washing over heated impervious surfaces (Galli 1991). Hawaiian streams in urbanized watersheds display greater daily temperature fluctuations than forested streams (Brasher 2003). Wang et al. (2003) calculated that stream temperature increases by 0.25 ºC with every 1% imperviousness. Temperature maxima in urban streams during low baseflow also pose a threat to stenylthermal biota (LeBlanc et al. 1997). Wehrly et al. (2003) found that fish community composition and species richness changed across temperature gradients with specific ranges and identified distinct cold, cool and warmwater assemblages. Therefore, temperature regime shifts may present a probable explanation for altered biotic assemblages in urban streams.

Evidence for urbanization-related changes in other stream parameters including pH and dissolved oxygen has not been clearly shown (Ragan and Dietemann 1975, Hatt et al. 2004). Pompeu et al. (2005) found much lower dissolved oxygen and slightly higher pH in urban Brazilian streams, however the low dissolved oxygen is most likely due to considerably high biological oxygen demand (BOD) from sewage discharge. Increased BOD has been shown in urban stormwater runoff (Ragan and Dietemann 1975), at levels similar to secondary wastewater effluent (Bryan 1971).

Sediment is a primary source of habitat and water quality degradation in urban streams (Waters 1995). Interestingly, urban stormflow runoff has been characterized by increased total suspended solids (Bryan 1971), yet Walters et al. (2003) found that urbanized highland streams in Georgia display high turbidity at baseflow levels as well. Geographic and soil type differences present a complex picture of stream sediment loads.

Australian streams indicated no significant relationship between impervious surface and total suspended solids (Hatt et al. 2004). Conversely, urban development was responsible for annual sediment yields 50% higher than in undeveloped Pacific NW watersheds, as a result of landslides, bank and road surface erosion (Nelson and Booth 2002). Although some variation does exist, there is enough evidence of sediment dynamics to justify a general positive relationship between urbanization and suspended sediment in streams.

One attribute of water quality that has been well documented in the urban literature is conductivity (Herlihy et al. 1998, Paul and Meyer 2001). Increased stream ion concentrations are a consequence of runoff over impervious surfaces, passage through pipes, and exposure to other anthropogenic infrastructure. Significantly increased conductivity has been shown in Australian (Hatt et al. 2004) and Georgia, USA (Rose 2002) urban streams. Chloride, specifically, has emerged as an important stressor to stream quality due to road de-icing (Kushal et al. 2005). Although it has been found in high levels in urban areas previously (Bryan 1971), the widespread use of salt to de-ice roadways in winter has led to regionally elevated chloride levels in stream water 25% higher than in seawater, remaining high throughout the summer even in less-impacted watersheds (Kushal et al. 2005). Thus, instream chloride levels may not be an indicator of localized urbanization, per se, but may reflect the results of regionalized road construction and land development.

Finally, recent USGS studies of urban streams across the US found elevated levels of detergent metabolites, steroids, plasticizers, non-prescription drugs, antibiotics and disinfectants as the six highest concentration wastewater components (Koplin et al. 2002). N,N-diethyl-m-toluamide, also known as DEET in insect repellent, was found in the highest concentration downstream from intense urbanization (Sandstrom et al. 2005). Thus, not only are urban streams subject to changes in water quality due to impervious surface runoff, but also due to the survival of these compounds through wastewater treatment plants (Fent et al. 2006).

Evidence of heavy metals has been shown in urban streams as well. Zinc, copper, cadmium, and lead concentrations increased with the percent imperviousness in urban Australian watersheds (Pettigrove and Hoffman 2003). Sediments from urban streams in Scotland exhibited concentrations of lead, copper, chromium, nickel and zinc above the allowed standards as well (Wilson et al. 2005). Therefore, heavy metal contamination is another common feature in urban systems due to runoff and industrial land use.

Ecosystem function

Stream ecosystem function involves chemical and physical processes that serve biotic communities. Leaf breakdown, production, respiration, ecosystem metabolism, and transformation of nutrients occur within the streambed, banks, and channel and all measures of ecosystem function. When these functions occur in a state of equilibrium, ecosystem services (benefits provided by natural ecosystem processes) supply terrestrial and instream biota with vitally essential products (Palmer et al. 2004). For example, instream breakdown of leaf litter into biologically available nutrients provides a foundation for the aquatic foodweb (Meyer et al. 2005). Nitrogen and phosphorous are two macronutrients that cycle through solute pathways, entering the system from upstream or terrestrial inputs, becoming suspended in the water column, retained in bars mid-channel, in the streambed, on the streambank or on the floodplain, taken up by biota, and exported to downstream receiving waters (Allan 1995). The cycle that nutrients pass through as they are transformed into an available nutrient, incorporated into living tissue, and returns to a dissolved, available form takes place over some distance of downstream transport (Allan 1995, Newbold et al. 1981). The shape of this cycle is thought to be a spiral and the length of one cycle can calculated, providing a measure of nutrient utilization (Newbold et al. 1981). Thus, information about nutrient processing offers an important picture of stream ecosystem functioning.

There have been few studies of ecosystem functioning in urban stream systems, although nutrient processing has been examined the most. Processing of inorganic nitrogen (nitrate) into organic forms is crucial to downstream ecosystems, due to the potential for eutrophication of coastal waters and contamination of drinking water by nitrate (USEPA 1990, Bowen and Valiela 2001, Boynton et al. 1996). Thus, a loss of denitrification zones and available carbon in urban stream systems has serious implications for the entire watershed (Groffman et al. 2005). Meyer et al. (2005) found that urban streams in Georgia, USA had higher instream nutrient levels due to increased inputs as well as reduced nutrient removal (longer spiral length) than forested streams. Interestingly, stream metabolism rates did not correspond to increased urbanization, yet leaf litter breakdown was negatively correlated to urbanization (Meyer et al. 2005). Retention of dissolved and particulate organic carbon decreases, yet their concentration has been shown to be higher in urban streams (Paul and Meyer 2001). In addition, Harbott and Grace (2005) found a positive correlation between the composition of dissolved organic carbon and the effective imperviousness within the watershed. Sources of carbon in urban systems thus reflect the qualities of stormflow runoff.

Organic debris jams in Maryland stream channels were found to exhibit higher denitrification rates in suburban streams than in forested streams, due to higher nitrate loading in urbanizing watersheds (Groffman et al. 2005). While this may seem to contradict the previous studies, Groffman et al. (2005) also argue that the debris jams may be a source of nitrate to downstream waters and that the lifespan of organic debris jams may be shorter in urban systems due to high storm flows. In support of this, research in desert southwestern US streams has shown that nutrient uptake (spiral) length is significantly longer in urban streams, thus maintaining higher nitrate concentrations throughout the stream network (Grimm et al. 2005). Therefore, retention of nitrogen (and thus transformation) in these streams was very low, providing limited biologically available nitrogen to downstream waters. Grimm et al. (2005) also relate these differences in ecosystem function to the lack of stream habitat complexity, e.g. presence of debris jams, in urban systems. In other biomes, suburban streams exhibited the highest levels of nitrogen retention compared to forested and urban streams, due to nearby lawn fertilizer sources (Groffman et al. 2004). Wahl et al. (1997) found nitrate concentrations were twice as high in urban streams than in forested streams, which was also correlated with greater annual streamflow volume. When urban baseflow and stormflow were compared, total dissolved nitrogen was significantly lower and dissolved organic carbon was higher during stormflow (Hook and Yeakley 2005).

Phosphorous, generally a limiting nutrient in aquatic systems, has been found in much higher concentrations in urban than in non-impacted streams (Paul and Meyer 2001, Brett et al. 2005, Hatt et al. 2004). Brett and colleagues (2005) discovered that urban streams had 95% higher total phosphorous and 122% higher soluble reactive phosphorous than forested streams. Sources of phosphorous in urban watersheds include fertilizers, wastewater effluent, and the soils' capacity to retain phosphorous in areas with a high density of septic tanks (LaValle 1975, Gerritse et al. 1995). Thus, evidence from studies of instream carbon, nitrogen, and phosphorous demonstrates that ecosystem function does appear to be altered in urban stream networks.

Riparian vegetation

Watersheds in rural, forested regions are characterized by intact riparian zones serving a variety of functions to the stream ecosystem. Trees, shrubs, and grasses that grow adjacent to the stream channel provide a natural filtration system for precipitation that is intercepted by the canopy, infiltrating the soils below. As water percolates through the soil, it is either taken up by vegetation, recharges the ground water, or is subsequently discharged into the stream channel laterally through the banks or from below through upwelling regions in riffles. In urbanized watersheds, riparian corridors are many times removed or narrowed along stream banks due to development of land adjacent to the channel. Buffer fragmentation due to housing and road construction decreases pollutant filtration and delivers increased sediment loads to the stream channel (Waters 1995). Patch dynamics within the riparian buffer zone change, decreasing the size of vegetation patches as the surrounding land becomes more urbanized (Aguiar and Ferreira 2005). Similarly, the function of riparian vegetation can be decreased when a stream is channelized, especially when lined by concrete. Groffman and colleagues (2003) measured water table depths and nutrient processing in riparian zones across an urban gradient that is currently being studied in the Baltimore Ecosystem Study. Monitoring indicated that urbanization generates hydrologic drought in riparian buffers, a condition in which the water table drops resulting in reduced function of the riparian vegetation and soil (Groffman et al. 2003). Previously hydric soils (saturated, commonly anaerobic conditions) become dry as a product of increased sediment deposition and lowered water table, reducing their capacity to perform denitrification (Groffman et al. 2003). Presence of stormwater pipes and road drains create shortcuts in the filtration path of precipitation (Paul and Meyer 2001). Instead of infiltrating through riparian soils, runoff (including pollutants) is sent directly to the stream channel. This modification in riparian buffer function reduces water quality drastically as mentioned above. In addition, riparian vegetation composition shifted from wetland species to more upland species in urban stream floodplains in comparison to forested floodplains (Brush et al. 1980, Groffman et al. 2003). Finally, loss of riparian canopy causes reduced large woody debris, which is important in structuring the stream channel and habitat within (Roy et al. 2005a). Therefore, urban land development plays a key role in shaping riparian vegetation composition, extent, and function.

Biotic communities

Urbanization impacts are particularly visible in many of the biotic components of the stream ecosystem. Historically, the effects of pollutants and degraded water quality were tested on various fish species, however more attention has been paid recently to biota found lower in the trophic food web. In addition, research on changes and/or loss of biotic communities has shifted from water quality effects to response of habitat loss and ecosystem services. The following pages will present research done on algae, diatoms, macroinvertebrates, fish, and other water-dependent vertebrates in urban systems.

Algae and diatoms

A few studies have examined changes in algal and diatom communities. Not surprisingly, urbanization affected the algal community composition in Massachusetts, Alabama, and Utah streams (Potapova et al. 2005). Urban streams were dominated by pollution-tolerant algal species, and changes in algal assemblages were associated with conductivity, nutrients, and physical habitat degradation (Potapova et al. 2005). There were geographic differences in algal composition, as well as differences in the component of urban streams the algae were responding to however.

Diatom communities have also been found to be good indicators of urbanization impacts. Newall and Walsh (2004) found a strong negative correlation between urbanization and diatom indices of water quality. They argue that high drainage connectedness results in the delivery of increased phosphorous and conductivity concentrations to streams, leading to changes in diatom community towards those species that indicate eutrophic conditions. Furthermore, shifts in diatom communities were directly linked to nutrient enrichment, providing another indicator of urbanization effects (Sonneman et al. 2001). Thus, algae and benthic diatom communities provide an important part of the biotic picture in urban watersheds.

Macroinvertebrates

Macroinvertebrate communities have been examined in many studies in response to land use change, including urban land development, and are severely degraded at low levels of urbanization and imperviousness. Stepenuck and colleagues (2002) found that levels of 8 to 12% connected imperviousness significantly decreases macroinvertebrate diversity in Wisconsin streams, while Morse and others (2003) found that streams in

Maine with 6% impervious cover exhibited abrupt changes in macroinvertebrate communities. Macroinvertebrate abundance is lower in urbanized stream reaches than forested reaches (Brasher 2003). Species richness generally declines with increasing percent of urban land use in the watershed (Gage et al. 2004, Roy et al. 2003, Walsh et al. 2001). Sensitive species, such as the Ephemeroptera, Plecoptera, and Trichoptera (EPT) insect group, are severely impacted by watershed urbanization. EPT richness was inversely correlated with the percent of urban land use in the watershed (Freeman and Schorr 2004, Roy et al. 2003, Stepenuck et al. 2002), demonstrating the lowest richness in highly impacted watersheds (Wang and Kanehl 2003, Gage et al. 2004). Subsequently, pollution-tolerant and introduced taxa were significantly higher in urban streams (Morse et al. 2003, Brasher 2003).

Effective stormwater drainage, increasing conductivity in receiving waters and other major water pollutants, is proposed as the cause of significantly increased abundance of a few tolerant taxa as compared to intolerant taxa in rural Australian watersheds (Walsh et al. 2001). In the southern U.S., urbanization increases sediment transport, total suspended solid concentrations, and decreases stream bottom substrate size resulting in decreased filter-feeders and predators (Freeman and Schorr 2004), low macroinvertebrate diversity, and increased numbers of tolerant species (Roy et al. 2003). Water quality was responsible for degradation in benthic communities in urban Michigan streams where industrial effluent was discharged; however, increased habitat quality through the generation of more riffle habitat during high discharge events enhanced specific functional groups of macroinvertebrates (Nedeau et al. 2003).

Structural habitat degradation has also been linked to urban effects on benthic invertebrate communities. Stream channels with intact riparian buffers had significantly greater diversity than those without buffers (Moore and Palmer 2005). Reach-scale channel characteristics such as slope and channel modifications were important in determining benthic community composition in streams in California (Fend et al. 2005), while in a study of multiple urban environmental settings, basin-scale variables were better predictors of the impacts of urbanization (Cuffney et al. 2005). Thus, although some geographic and scale differences occur in response to urbanization, stream macroinvertebrates communities change with low levels of watershed urbanization, presenting high abundances of intolerant species, and low abundances and richness of sensitive species.

Fish

In comparison to less-impacted systems, urban streams maintain fish assemblages characterized primarily by warmwater, pollution-tolerant omnivores and generalists (Pirhalla 2004, Kemp and Spotila 1997, Morgan and Cushman 2005, Schweizer and Matlack 2005, Roy et al. 2005b, Walters et al. 2005). In Maryland streams, blacknose dace *Rhinichthys atratulus* was found to be the dominant urban fish species (Klein 1979, Morgan and Cushman 2005). Blacknose dace is considered extremely tolerant of environmental conditions (Pirhalla 2004). Pollution intolerant fish species, such as brown trout *Salmo trutta*, are absent in urban streams, while dominating less-impacted headwater systems (Kemp and Spotila 1997). Rosyside dace *Clinostomus funduloides* was not found in urban streams in the 1970's after historical data showed great abundance in Maryland (Ragan and Dietemann 1975). After monitoring an urbanizing stream for 8 y, Schweizer and Matlack (2005) found that urban fish assemblages were dominated by high silt tolerant species and lost fish species preferring gravel substrate. Interestingly, changes in the fish assemblage occurred prior to major changes in stream physical habitat.

Species richness also decreases as the amount of urban land use in the watershed increases (Morgan and Cushman 2005, Paul and Meyer 2001, Weaver and Garman 1994). In some stream networks, fish richness and abundance decreased downstream as the intensity of urbanization increased, yet other characteristics of the fish assemblage increased as river conditions near the mouth improved (Tabit and Johnson 2002). Examples like this are important to discuss because the effects of urban land use on small streams are much more severe than in larger, higher order streams. Changes in food web structure associated with urbanization were found to be the cause of diet shifts in many fish species (Poff and Allan 1995, Weaver and Garman 1994).

Indices of biotic integrity (IBI; Karr 1981) have been widely used to illustrate the impacts of urbanization and other changes in land use/ land cover on fish assemblages. An IBI is a summary of metrics that describes the health or condition of a biotic community, many times used as a management tool to compare streams and watersheds by their rank or score. Urbanization was negatively correlated with fish IBI scores in Wisconsin, Ohio, Maryland, North Carolina, Tennessee and Georgia streams (Long and Schorr 2005, Morgan and Cushman 2005, Helms et al. 2005, Kennen et al. 2005, Miltner et al. 2004, Roth et al. 1998, Volstad et al. 2003, Wang et al. 2003). Different geographic locations and assemblage composition govern the threshold at which IBI scores drop when correlated to percent impervious surface. Minor changes in fish assemblages were found between 6 and 11% impervious surface in Wisconsin trout streams (Wang et al. 2003), yet studies of other streams in Wisconsin indicate a threshold between 10 and 20% impervious surface, but 8 to 12% connected impervious surface (Wang et al. 2001, 1997). Ohio streams presented significantly lower IBI scores over 14% impervious surface. Morgan and Cushman (2005) documented that eastern Piedmont streams in Maryland with greater than 25% urban land use were classified as having poor biotic health. Thus, depending on the resident fish assemblage and the age of urban development in the watershed, poor biotic integrity may be found at very low levels of anthropogenic impact.

Urbanization has been shown to affect the vital rates of impacted fish species. Urban blacknose dace experienced increased growth rates during their first year of life when compared to dace in rural streams (Fraker et al. 2002). Yet in heavily urbanized watersheds (>90% urban land use), blacknose dace were smaller and younger at maturity due to a greater percentage of the population mature at age one (Fraker et al. 2002). Conversely, urbanization effects produced higher biomass but changed the age structure of salmonid populations in Washington. Urban fish populations consisted of more age 0 and I fish than a more diverse age structure and species assemblage in rural streams (Scott et al. 1986). Limburg and Schmidt (1990) were the first to reveal a significant negative relationship between urbanization and egg and larval densities of anadromous fish in Hudson River tributaries. Thus, there is evidence that urbanization impacts may play a role in shaping the life history and population ecology of fishes (Schlosser 1991).

Road and sewerline crossings are detrimental to fish habitat and population dynamics in urban stream ecosystems. In particular, Warren and Pardew (1998) found that movement of centrarchids, cyprinids, and fundulids through culverts was an order of magnitude lower than other types of road crossings or through natural stream reaches. Additionally, fish assemblage richness and biomass was significantly lower above sewerline crossings as compared to assemblages below sewerlines in the Ohio River Valley (Koryak et al. 2001).

The term homogenization has recently been used as an indicator of the long-term effects of urbanization on fish assemblages and refers to the ratio of endemic to cosmopolitan species (McKinney 2006, Scott 2006, Roy et al. 2005b, Walters et al. 2003, Rahel 2002). Loss of native species and invasion of non-natives may result in homogenization in urban systems, however Marchetti et al. (2006) described differentiated fish assemblages due to varying rates of invasion and endangerment. Walters et al. (2003) and Roy et al. (2005b) argue that urbanization in Georgia streams caused homogenized fish assemblages due to physical stream conditions, such as silt and stormflow tolerance, that favor cosmopolitan species. Similarly, Scott (2006) used the difference between endemic and cosmopolitan species to signify homogenization and concluded that endemic species "lose out" while cosmopolitan species "win" along the urbanization gradient. Due to its frequency of use recently, this indicator may become a valuable tool in assessing the impacts of land use changes on fish assemblages.

Other water dependent vertebrates

There is some evidence that urbanization has significant impacts on higher vertebrates that spend their life partially in stream networks. Many species of Californian frogs and newts have been observed at low abundances in urban streams, responding to very low levels of % urban land development, similar to studies of fish and benthic macroinvertebrate assemblages (Riley et al. 2005). Changes in biotic composition were likely due to degraded physical habitat such as the number of pools, on which amphibians rely. Bowles et al. (2006) examined the distribution of *Eurycea tonkawae*, a salamander found in Texas springs and caves, and found decreased densities in developed watersheds, where high conductivity was also measured. In Australia, platypus populations were only located at stream sites with less than 11% imperviousness, indicating that these animals are also extremely sensitive species to anthropogenic change (Serena and Pettigrove 2005). These impacts may be due to indirect effects of prey availability or the tolerance of vertebrates to water quality and specific stream habitat.

Experimental approaches to studying urbanization

There are many general approaches to document change in a natural environment. One method is to use the BACI – the Before-After-Control-Impact which is used to separate anthropogenic effects from other variability in space and time (Green 1979). This design involves two conditions or streams, one of which receives some type of change (impact) while the other remains unchanged (control) and are compared prior to and post-impact. A second approach to determine if a factor affects a response is to experimentally manipulate one component of a system and compare the results to other treatment responses in other streams. I used this approach in the third chapter to better understand habitat patch selection by fish in urban, suburban, and rural streams. A paired watershed design is third method used in which prior (e.g. anthropogenic) change within the watershed is predicted to elicit differential responses in a set of parameters. This has more commonly been used in hydrological studies, but appears within studies of stream geomorphology as well (Roy et al. 2005a, Pizzuto et al. 2000, Burges et al. 1998). I used this experimental approach in my forth chapter to examine if differential fish movement patterns existed in urban and rural streams. A forth approach to study the effects of urbanization is through comparisons along an urban to rural gradient (Limburg and Schmidt 1990, McDonnell and Pickett 1990, Morgan and Cushman 2005, Fraker et al. 2002). McDonnell and Pickett (1990) were the first to declare this gradient as a natural experimental framework to study and understand the effects of urban land use. One problem that arises when trying to characterize and understand differences between impacted and non-impacted systems is that there are few places that have been able to avoid anthropogenic influence in some manner. It is very difficult to find a 'pristine' area along the east coast of the US, which makes identifying a 'control' for field studies almost impossible. However, scientists have begun to acknowledge and use a landuse continuum model that includes human influence to study the effects of habitat fragmentation and alteration, changes in species richness and abundance, and ecosystem services in aquatic environments (Theobald 2004, McDonnell and Pickett 1990, Morgan and Cushman 2005, Collins et al. 2000, Fraker et al. 2002). The use of landuse or landcover gradients within a watershed provides a framework for the study of subtle changes in ecological function and structure, and therefore forms the basis for my study of physical habitat within small streams (Chapter 2).

Conclusions

Urban stream ecosystems are especially in need of remediation due to overall degradation of structure and function. The ability to describe, model, and predict the future of freshwater stream systems may provide an advanced understanding of pollution tolerances and limitations. Sound environmental policymaking requires solid science to inform decisions made to protect, conserve, and restore natural resources. Models that predict biological patterns and interactions over a range of environmental conditions are useful tools for resource managers and policymakers to agree on the extent that we can alter a system from its "original" state without causing community collapse. These tools could also promote cost effectiveness by identifying the areas that need restoration and preservation the most. Cultural traditions have and continue to make fish populations very important to our society both commercially and recreationally. Therefore, understanding the persistence and dynamics of fish communities in degraded habitats should be of concern to all.

Indices of biotic integrity and other new strategies to identify the conditions of aquatic health have greatly enhanced our ability to prioritize our resources and efforts, and understand the ecological challenges we are presented with as a human-dominated ecosystem (Vitousek et al. 1997). However, these metrics do not decipher why communities are structured in a particular way. Ultimately, knowledge of the mechanisms that shape the structure of these fish assemblages, and the thresholds exhibited by certain fish species, would increase our understanding of biotic and abiotic interactions in highly degraded ecosystems. The following research examines relationships between stream habitat dynamics and fish assemblages across an urban – rural gradient to provide a framework on which to direct further studies of urban ecology.

As indicated by the recent growth of urban stream studies (Figure 1) and resultant wealth of knowledge, an understanding of land use change impacts is crucial to our preservation, conservation, and restoration of ecosystem structure and function in the future. It remains important to examine how systems respond in different geographic regions due to the diverse patterns seen in chemical, physical, and biological aspects of urban watersheds. The literature base has grown significantly over the last two decades on stream networks that span the globe, symptomatic of increasing anthropogenic stress from an expanding human population. However, there are still gaps in our understanding. Particularly in Maryland, information about fish species and assemblage response to watershed urbanization was unknown. Therefore, a study using the Maryland Biological Stream Survey (MBSS) was designed to assess the changes in fish assemblage patterns in small streams in the eastern Piedmont and Coastal Plain of Maryland, which is presented in the following chapter. Research on how fish populations respond to not only habitat degradation, but also to the restoration of stream channels is scarce. The impact of urbanization on fish communities in streams across Maryland is severe (Klein 1979, Morgan and Cushman 2005, Roth et al. 1999), yet mechanisms which relate habitat use and fish movement, and thus assemblage structure in urban streams have not yet been evaluated. Therefore, I designed three studies (Chapters 3 to 5) to link current gaps in the ecological and environmental knowledge of urban stream fish assemblages with stream quality and their habitat use.

Overview of hypotheses and following chapters

To appropriately assess the impacts of urbanization on the fish assemblages in the eastern Piedmont and Coastal Plain of Maryland, I used the MBSS statewide database to compare species richness and abundance in 1^{st}, 2^{nd}, and 3^{rd} order streams. Data collected at sites selected for this study (n = 544) were used to answer the following questions. Do relationships exist in varying stream orders between urbanization and: 1) fish abundance, 2) species richness, 3) fish index of biotic integrity (FIBI), 4) difference between expected and observed fish assemblage patterns? It was hypothesized that abundance, species richness, and FIBI would decline with urbanization in both the eastern Piedmont and the Coastal Plain.

Although many studies have documented differences in physical stream habitat in paired watershed studies, I know of no studies that attempt to document change across the urban-rural gradient. The third chapter examines the characteristics and complexity of instream and streambank habitat to determine what changes occur across the urban – rural land use gradient. This study incorporates data collected at over 50 stream sites spanning the Baltimore-Washington corridor with the percent urban land use in the watershed ranging from 0 to 80%. I examined these stream sites to determine if stream habitat quality changes as a function of urbanization. Specifically, I hypothesized that 1) variability in channel morphology and subunits change, 2) the drainage connection between stormwater drains and stream channels influences the extent of erosion and bar substrate size, 3) water quality declines, and 4) the quantity of good instream habitat declines. I expected that some but not all measures of stream habitat quality will change significantly across the urban-rural land use gradient, indicative of the variability in urban stream degradation. In addition, relationships between urban land use and characteristics of habitat use may or may not be linear. However, outcomes of this study will be useful in assessing what components of stream habitat are impacted the most from urbanization as well as how developed a watershed may become before changes occur within the physical streamscape.

As physical habitat changes across the urban-rural gradient, habitat preference or use by fish assemblages may also change. Chapter 4 focuses on a short-term response of fish to experimental enhancement of instream habitat patch complexity. The motivation for conducting this study comes from recent evidence of biotic response to stream restoration practices. As a result of few studies demonstrating improvements in biotic communities following habitat restoration practices in degraded streams, I designed an instream experiment in which fish habitat was manipulated and habitat selection response was measured. This experiment was conducted with three enhancement treatments (large woody debris, shade or both) in urban, suburban, and rural streams. Would urban, suburban, and rural fish assemblages respond similarly to habitat restoration? Given the choice of enhanced versus unenhanced habitat within each stream site, I hypothesized that fish would select the enhanced habitat greater than 50% of the time in all stream/land use categories. Specifically, I hypothesized that 1) fish in rural streams would select shade or combined shade and large woody debris more than just woody debris, 2) fish in suburban streams would respond better to a combination of large woody debris and shade than other types of enhancement, and that 3) urban fish would not select any one enhancement more than another. From these experimental manipulations, I expected that fish response would vary by landuse category, as well as by enhancement treatment applied. Results from this study may provide a better understanding of differential fish response to stream habitat restoration on a short-term basis.

Available habitat and fish preference may also imply differential movement patterns and home range between urban and rural stream ecosystems due to environmental and ecological stress. In the fifth chapter, I present a movement study of two stream cyprinid species in rural and urban streams. Recent literature suggests that some fish populations may be split into mobile "movers" and sedentary "stayers" groups due to ecological influences on fish behavior. However, I hypothesized that the proportion of movers and stayers differs between urban and rural streams due to environmental and stream habitat differences. Thus, I also hypothesized that stream habitat differences are evident between urban and rural streams. Finally, I hypothesized that urban fish exhibit greater home ranges than rural fish because of a poor local resource base and living in a highly disturbed environment. I expected that urban fish populations would display a greater mover subpopulation, using large expanses of a stream reach. Since urban streams experience a higher intensity of disturbance from altered hydrological patterns, fish may also become displaced after high stormflows. Results from this study provide key information on important ecological interactions in degraded stream ecosystems as well as life history variation.

In conclusion, the final chapter summarizes the results from each of these studies, indicating the most significant results from each chapter. I provide some closing thoughts about the conclusions and implications of this research, addressing questions that were unanswered by these field studies. Finally, I leave the reader with some direction as to what further research is required to fill gaps in understanding the ecological and environmental aspects of degraded urban stream ecosystems.

Figures

Figure 1. The number of studies involving urban stream ecosystems from 1970 to 2006.

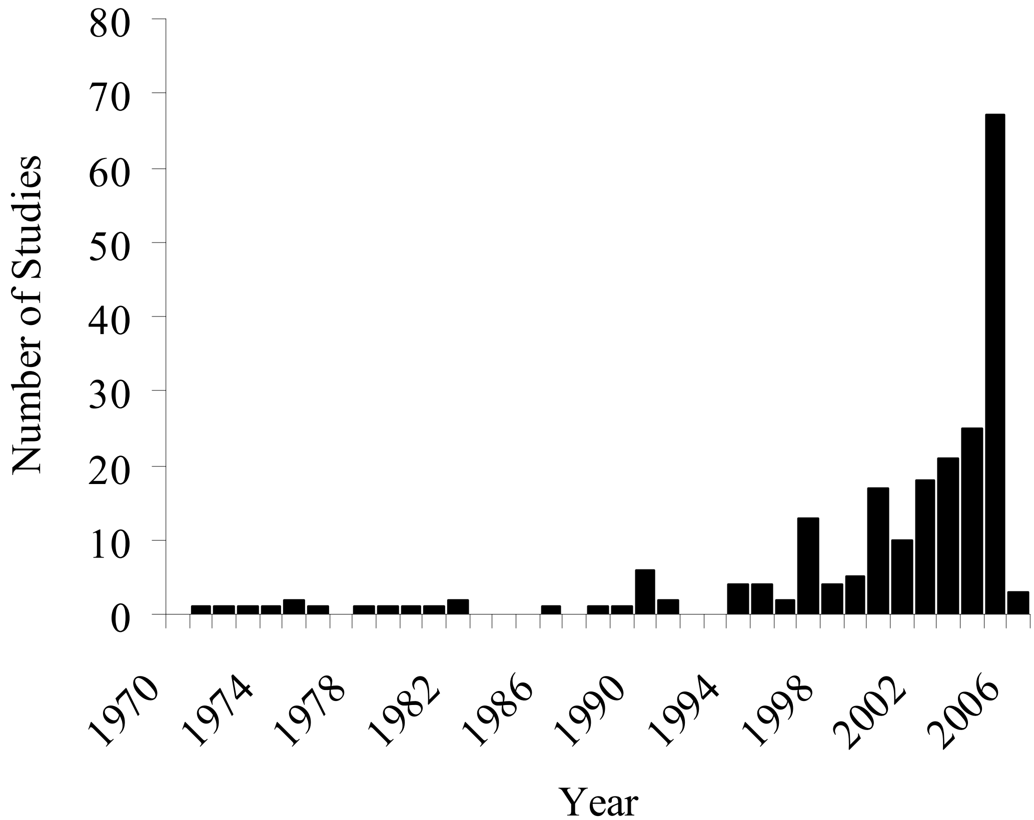
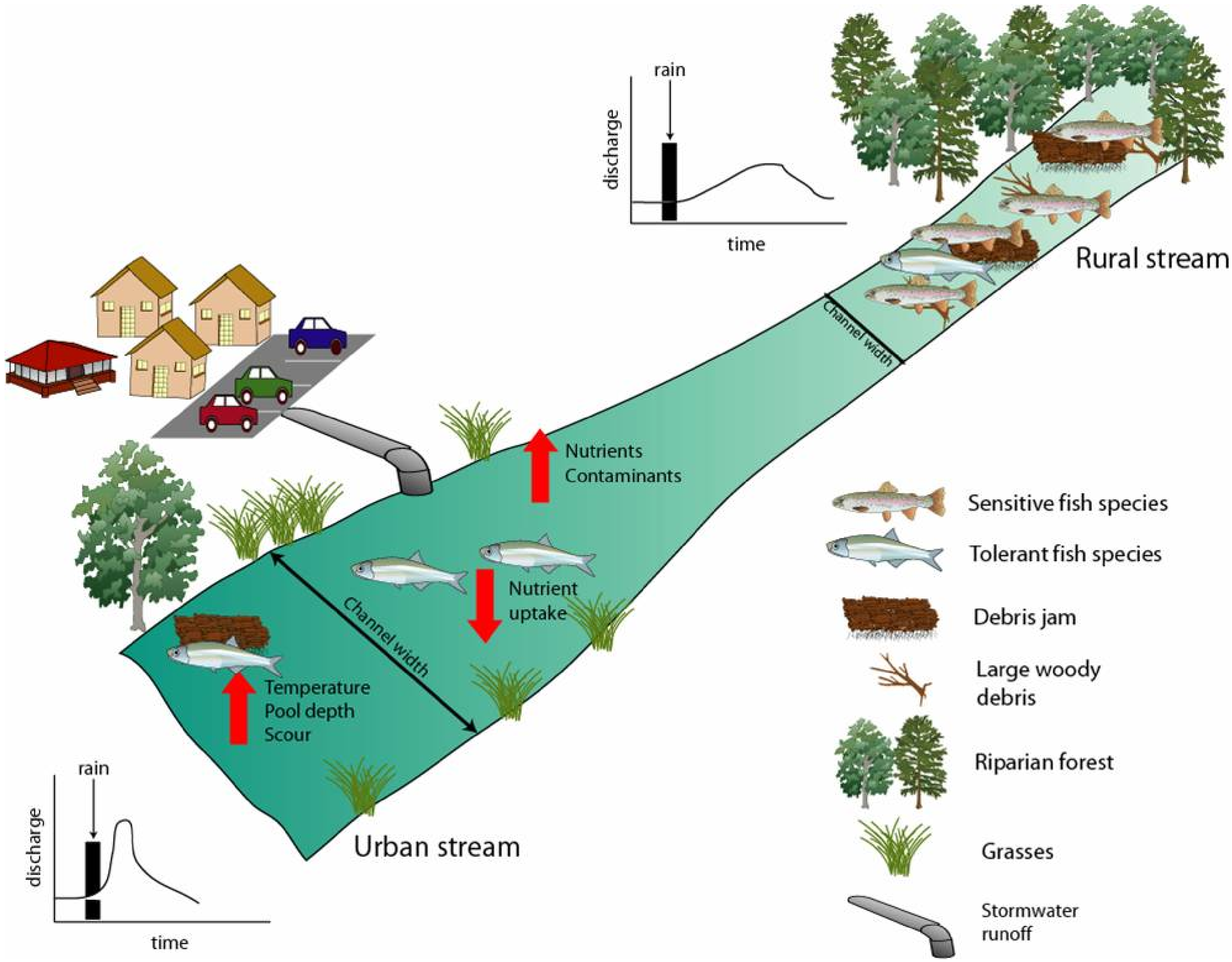


Figure 2. Symptoms of degradation commonly found in small urban streams, referred to as the urban stream syndrome. When compared to rural streams, urban streams display changes in hydrology due to connected impervious surfaces and stormwater runoff, nutrient processing, channel habitat complexity and morphology, and riparian vegetation composition.


rain
discharge
time
Rural stream
Channel width
Nutrients
Contaminants
Sensitive fish species
Tolerant fish species
Channel width
Nutrient
uptake
Debris jam
Temperature
Pool depth
Scour
Large woody
debris
Riparian forest
rain
discharge
time
Urban stream
Grasses
Stormwater
runoff

Chapter 2: Urbanization effects on stream fish assemblages in Maryland, USA.

Abstract

We examined patterns in Maryland fish assemblages in 1^{st}- through 3^{rd}-order nontidal sites along an urbanization gradient in the eastern Piedmont (EP) and Coastal Plain (CP) physiographic ecoregions of Maryland, USA, using 1995 to 1997 and 2000 to 2002 data from the Maryland Biological Site Survey (MBSS). Major urbanization and other historical stressors occur in both ecoregions, and there is potential for further stress over the next 25 y as urbanization increases. We assigned each MBSS site (n = 544 streams) to a class of urbanization based on land cover within its upstream catchment. We compared observed fish abundance and species richness to the probable (expected) assemblages within each ecoregion, and also assessed the accuracy of the Maryland fish index of biotic integrity (FIBI) to indicate catchment urbanization. Relationships between urbanization and fish assemblages and FIBI varied between the 2 ecoregions. Assemblages in EP streams exhibited stronger relationships with urbanization than those in CP streams, particularly when urban land cover was >25% of the catchment. Across all EP stream orders (1^{st}, 2^{nd}, and 3^{rd}), high urbanization was associated with low fish abundance and richness, low FIBI, and few intolerant fish species, resulting in assemblages dominated by tolerant species. Conservation practices minimizing urbanization effects on fish assemblages may be inadequate to protect sensitive fish species because of the invasiveness of urban development and stressors related to the urban stream syndrome.

Introduction

The "urban stream syndrome" prevails when human population density reaches a critical limit within a catchment (Paul and Meyer 2001, Groffman et al. 2005, Meyer et al. 2005). Such modification in stream structure and function often results in degraded physiochemical conditions and associated changes in biota (Paul and Meyer 2001, Roth et al. 1999, Gergel et al. 2002, Meyer et al. 2005, Walsh et al. 2005a).

Effects of urbanization on stream communities have been reported worldwide (Forman and Alexander 1998, Paul and Meyer 2001, Forman et al. 2003, Walsh et al. 2005b). Paul and Meyer (2001) noted that urbanization is second only to agriculture as an agent of stream degradation in the US (see also USEPA 2000). Once catchments are urbanized, intermittent and perennial streams may show altered hydrologic regimes, elevated nutrient and contaminant concentrations, and degraded biota, which may be difficult to mediate or reverse (Booth 2005, Paul and Meyer 2001, Groffman et al. 2003).

Numerous studies have reported that changes in catchment land use affects stream fish populations (e.g., Pirhalla 2004, Fraker et al. 2002), although few studies have documented fish assemblage responses to urbanization (reviewed by Paul and Meyer 2001). The 6 studies cited in Paul and Meyer (2001) generally found changes in either fish diversity or indices of biotic integrity with increasing catchment imperviousness, with changes typically occurring at 10 to 12% imperviousness (e.g., Klein 1979, Steedman 1988, Wang et al. 1997, Yoder et al. 1999).

Fish assemblages in small (1^{st}- to 3^{rd}-order) perennial streams are particularly at risk from urbanization impacts. These streams often exhibit naturally low fish richness, and thus are highly susceptible to loss of species and overall diversity from urbanizationinduced changes in water quality, hydrologic regimes, or both. In addition, the relatively close proximity of land use changes to small streams may have harsh, immediate effects on fish assemblages including loss of breeding, feeding, and resting habitat (Paul and Meyer 2001, Bunn and Arthington 2002). In many areas, housing developments and individual home sites, are increasingly invading previously forested or farmed headwater catchments, often far upstream of urban centers. Within a catchment, headwater fish assemblages also may become isolated from downstream source populations by downstream barriers in urban channels (e.g., impoundments; Pringle et al. 2000).

Urbanization is an acute problem within Maryland, USA, especially along the Baltimore–Washington, DC corridor. Maryland's human population increased from 3.9 to 5.3 million from 1970 to 2000, with a projected increase to 6.3 million by 2025 (Maryland Department of Planning 2002: www.mdp.state.md.us/msdc/popproj). The urban stream syndrome is not new to the state, with urbanization impacts dating to at least 1790 (Otterstrom 2003), which, along with early agriculture, has shaped freshwater communities. Two mid-Atlantic ecoregions in particular, the eastern Piedmont (EP) and Coastal Plain (CP), have the highest population density in the state (4–14 people/ha; Roth et al. 1999), and recently have experienced drastic increases in forest fragmentation and forest cover loss. It is likely, therefore, that instream biotic conditions and processes, including fish assemblages, have been highly degraded in these ecoregions (Griffith et al. 2003).

We quantified relationships between catchment urbanization and stream fish assemblages in the EP and CP ecoregions of Maryland. In addition, we also assessed whether urbanization-assemblage patterns in each ecoregion varied among 1st-, 2nd-, and 3rd-order streams draining catchments with contrasting urbanization.

Methods

Study area and data source

We examined patterns between urbanization and fish assemblages in EP and CP (Fig. 1) using the Maryland Biological Stream Survey (MBSS) data base. This statewide stream survey was conducted by the Maryland Department of Natural Resources (MDDNR), Versar, Inc., and the University of Maryland between 1995 to 1997 (Round 1) and 2000 to 2002 (Round 2, continued through 2004). Initially, MBSS was designed to assess impacts of acidic deposition and anthropogenic impacts on stream biotic integrity of fish and benthic macroinvertebrates within specific biogeographic regions (Kazyak 2000, Roth et al. 1999).

MBSS is a hierarchical probability-based survey that was focused on small streams (Heimbuck et al. 1999, Roth et al. 1999). Round 1 sampling was conducted on wadeable 1st- through 3rd-order nontidal streams, composing 89% of the total stream length in Maryland (Roth et al. 1999). Each sampling site was randomly generated using a Geographical Information System (GIS, 1:250,000 scale) that incorporated statewide stream network information, but kept the total number of sites proportional to the number of stream km within a given order (Heimbuck et al. 1999, Roth et al. 1999). MBSS field crews used Global Positioning System (GPS) coordinates of each site to locate the middle of the sampling segment, and a 75-m reach per site was measured and marked (Kazyak 2000, Roth et al. 1999).

Statewide, mean stream width (m) and thalweg depth (cm) ranged, respectively, from 2.3 and 16.8 for 1st-order streams to 8.8 and 41.8 for 3rd-order streams (Roth et al. 1999). Mean summer discharge in 1st-, 2nd-, and 3rd-order streams was 0.023, 0.13, and 0.36 m^3/s respectively (Roth et al. 1999).

Catchment classification

MDDNR personnel quantified land use within the upstream catchment of each MBSS site using GIS (1:62,500 scale) and landuse/landcover data (Federal EPA Region III Multi-Resolution Land Characteristics, 30 × 30 m resolution). All catchments with >65% agricultural land use were eliminated to reduce confounding effects of current agriculture on fish assemblages; however, this approach did not account for historical agricultural influences. A total of 544 MBSS sites met all criteria for site selection and comprised the primary data set for the analysis (i.e., 265 EP and 279 CP sites, Table 1; Fig. 1). We classified the resulting study sites into discrete groups based on the % urban land cover in the catchment (= % catchment urbanized).

Fish sampling

MBSS conducted fish surveys during summer (1 June–30 September 1995–1997 and 2000-2002) using electroshockers (Model 12; Smith-Root® Inc, Vancouver, Washington) and block nets placed at the upstream and downstream ends of the 75-m sampling reach; fishes were collected using the double-pass method (Heimbuck et al. 1997). Abundance (no. of individuals/site) and species richness (no. of species/site) were recorded at each site. In addition, baseflow discharge, several instream physical habitat parameters (i.e., stream alteration, bank erosion potential, instream habitat structure quality and quantity, and stream channel subunit dimensions), and riparian buffer widths were quantified using methods in Kazyak (2000).

Data analyses

We asked 3 questions using the EP and CP fish data sets. First, do relationships between catchment urbanization and fish abundance, and urbanization and species richness, vary with stream order? Second, are measures of assemblage biotic integrity (Roth et al. 1998, 2000) sensitive to urbanization, and do these relationships vary with stream order? Third, do observed relationships between fish assemblages and urbanization differ from expected or probable patterns, and do differences between observed and expected patterns vary with stream order?

We addressed the above questions using several statistical analyses. Random assignment of MBSS sites by year yielded low sample sizes in several urbanization categories, especially for EP sites in the 10–25% urbanized category (Table 1). Therefore, we combined EP sites into 3 groups (0–25%, 25–50%, and >50% of catchment urbanized), whereas for CP sites, we divided the 1st- and 2nd-order sites into 4 groups (0–10%, 10–25%, 25–50%, and >50% urbanized), and 3rd-order sites into 2 groups (0–25% and >25%). We compared abundance and richness for each urbanization category against the lowest urban level using ANOVA. If significant differences occurred, we used a Least Significant Difference (LSD, Steel and Torrie 1960) test to determine which group differed from the least-urbanized group. We used Levene's test (Levene 1960) to assess homogeneity of variances and, if data were nonnormal, we log-transformed them ($\log_{10}[x+1]$) prior to analysis (Zar 1974).

We tested the degree to which measures of assemblage biotic integrity corresponded with urbanization by regressing % of catchment urbanization against the Maryland fish index of biotic integrity (FIBI, Roth et al. 1999). We used regression instead of ANOVA here because use of ANOVA for multimetric indices, such as FIBI, is considered inappropriate (Norris and Georges 1993). FIBI values range from 1 to 5, where 1.0–1.9 is considered "very poor", 2.0–2.9 "poor", 3.0–3.9 "fair", and 4.0–5.0 "good" (Roth et al. 1998, 1999, 2000).

We assessed differences between expected and observed species richness for each ecoregion to estimate potential species loss associated with urbanization. EP and CP richness were based on the 16 probable (expected) Maryland stream assemblages from the MBSS data set, as derived by Kilian (2004), using clustering techniques that determined fish assemblages based on similarities in species composition (constancy) and relative abundance (Table 2). We used these groupings to define the probable fish assemblages that should occur in each ecoregion and stream order, to which we compared observed fish assemblages. We determined observed richness by the presence of an individual of each species per MBSS site for each ecoregion, and used X^2 to test if observed and expected richness differed at each site. Subsequently, we artificially lowered the expectations of richness in the species complex incrementally by one species to determine when observed assemblages in all urban categories departed significantly from the new expected assemblage. We used the MBSS intolerant and tolerant fish species designations from Roth et al. (1998, 2000) for EP and CP sites; these designations generally corresponded to tolerance values of McCormick et al. (2001) and Pirhalla (2004). We set significance for all statistical tests at $\alpha = 0.05$ (Steel and Torrie 1960).

Results

For both CP (Table 3) and EP (Table 4), fish richness and abundance in sites at the lowest urbanization level increased with increasing site order. As catchment urbanization increased, richness in EP sites also decreased within each order (Table 4), whereas richness in CP sites did not (Table 3). Similar to richness, fish abundance increased at the lowest urbanization level as site order increased in both ecoregions (Tables 3, 4); however, there was a general decline in abundance in EP sites within each order as catchment urbanization increased (Table 4).

CP patterns

1st-order streams—Mean fish species richness ranged from ~4 to 6 per site across all urbanization categories (Table 3). There were no significant differences in fish abundance or richness across all urbanization levels (Tables 3, 5). Abundance in highly urbanized sites was only slightly lower than the least-urbanized sites. Slightly higher fish abundances in 0–10% and 10–25% than >50% urbanized sites resulted from increased presence of tolerant fish species and an overall reduction of species in other tolerance categories (Table 2).

2nd-order streams—Mean richness ranged from 11 to 12 species per site across all urbanization categories (Table 3). Abundance and richness did not significantly differ among urbanization levels (Tables 3, 5); however, high abundances of fish per site at the 2 highest urbanization levels (>330 fish per site; Table 3) was possibly associated with replacement of intolerant with tolerant species (generalists) as catchment urbanization increased.

3rd-order streams—Mean richness ranged from 15–16 species across both urbanization categories (Table 3). Mean richness and abundance was higher in 3rd-order sites than in 1st-and 2nd-order sites (Table 3). Neither fish abundance nor richness differed between the 2 urbanization levels for 3rd-order sites (Tables 3, 5).

EP patterns

1st-order streams—Mean richness ranged from ~3 to ~7 species per site across all urbanization categories (Table 4), and richness significantly differed among urbanization categories (Tables 4, 5). Richness was generally low in sites with >25% urbanization, rarely exceeding 5 species per site, whereas richness in >50% urbanized catchments was <3 species per site. Fish abundance in sites from >25% urbanized catchments was significantly lower than in less-urbanized sites (Tables 4, 5). Mean abundance in the 0–25% urbanized sites was ~2.5 times higher than >50% urbanized sites (Table 4).

2nd-order streams—Mean richness ranged from ~5 to 12 species across all urbanization categories, with a progressive decrease from the least- (0–25%) to the most-urbanized (>50%) catchments (Table 4). Richness significantly differed between the least- and most-urbanized catchments (0–25% vs. >50%, respectively; Tables 4, 5). Abundance did not differ among urbanization levels (Tables 4, 5); however, high abundance of fish in sites with >50% urbanization resulted from high numbers of tolerant blacknose dace (*Rhinichthys atratulus*). More than 1400 *R. atratulus* (98% of fish collected) were collected in 1 highly urbanized catchment (>75% urbanization), and >200 *R. atratulus* per site were found in 3 other catchments with >75% urbanization.

3rd-order streams—Mean richness ranged from ~5 to 17 species across all urbanization categories (Table 4). Richness values were significantly different among urbanization categories (Tables 4, 5), and rarely exceeded 5 species per site in >50% urbanized catchments. Richness differed significantly among urbanization categories, where 0–25% urbanized sites displayed fish richness >3 times higher than in the >50% urbanized sites and 2.5 times higher in the 25–50% urbanized sites (Tables 4, 5). Abundance also differed with degree of urbanization, being ~1.8 times and >3 times lower in the 25–50% and >50% urbanized sites, respectively than in the 0–25% urbanized sites (Tables 4, 5).

FIBI patterns

In CP sites, FIBI was inversely correlated with catchment urbanization ($p < 0.05$; Fig. 2A), although fit to the regression line (not shown in figure) was extremely low (r^2 = 0.035) because of high intersite variation. In contrast, EP sites displayed a strong inverse relationship between FIBI and % catchment urbanization ($r^2 = 0.49$, $p < 0.0001$, Fig. 2B). Using the breakpoint of 3.0 that separated "poor" from "fair" FIBI scores (Roth et al. 1999), we estimated that >20 and >29% catchment urbanization within CP and EP sites, respectively, could result in either a "poor" or "very poor" FIBI rating.

Species assemblages

Based on Kilian's (2004) species complexes (Table 2), 1st-, 2nd-, and 3rd-order CP sites were expected to have 8, 10, and 16 species (Table 6), respectively, whereas 1st-, 2nd-, and 3rd-order EP sites were expected to have 4, 10, and 8 species, respectively (Table 7). For CP sites, there were significant differences between expected and observed species assemblages at all levels of urbanization. When richness model expectations were lowered, differences between observed and expected assemblage did not become nonsignificant across all urbanization categories until the expected species number was 2 for 1st-order sites (25% of the expected species assemblage), 6 for 2nd-order sites (60%), and 9 for 3rd-order sites (60%; Table 6). Interestingly, comparison of observed and expected richness values indicated that assemblages differed in richness and in composition in CP sites (Tables 2, 3). Specifically, 2nd-order sites displayed higher observed species richness than expected, 11.5 vs. 10 respectively (Tables 3, 6); however, the species composition found at these sites differed from the expected assemblage. These results were unique because first order sites exhibited lower richness, and third order sites showed a similar richness to what was expected (Tables 3, 6).

For EP sites, there were significant differences between expected and observed species assemblages at >50% urbanization for 1st- and 3rd-order sites, and at >25–50% urbanization for 2nd-order sites. Differences between expected and observed assemblages did not become nonsignificant for all urbanization levels until expected richness was lowered to 3 for 1st-order sites (75% of the species assemblage), 5 for 2nd-order sites (50%), and 5 for 3rd-order sites (63%) (Table 7). Differences in fish assemblages for EP sites (Table 7) were usually observed at higher levels of urbanization (>50%) than CP sites (Table 6).

<u>*Discussion*</u>

Fish assemblages and FIBI

Using the MBSS data set, we found that Maryland stream fish assemblages were associated with urban land use, with major assemblage differences generally occurring at >25% catchment urbanization. Yet, our analyses showed strikingly different patterns in the 2 ecoregions. Neither abundance nor species richness differed between streams in low- vs. highly urbanized catchments in CP, whereas in EP streams abundance, richness, and FIBI all decreased with increasing urbanization. Moreover, richness and abundance decreased in 1st-, 2nd-, and 3rd-order EP sites as catchment urbanization increased, except for elevated abundance of tolerant species in 2nd-order EP sites. We found no evidence for a similar trend in CP sites, where fish assemblage composition apparently shifted from the complex expected to one that was unresponsive to urbanization. The probable assemblage was derived from fish occurrences across the entire CP instead of just the western shore of the CP, but fish richness and abundance did not change as urban intensity increased. Furthermore, the expected assemblage in CP sites was dominated by more tolerant species than sites in the EP, even at low-urbanization levels.

The significant negative relationship in the FIBI for EP sites but not CP sites with increasing urbanization was interesting because the FIBI was developed specifically for each ecoregion (Roth et al. 1999). However, our results suggest that components of the FIBI are useful in understanding potential fish response to urbanization in EP but have limited application in CP.

Richness, abundance, and FIBI provided limited information about fish assemblage–urbanization relationships in CP sites, but we found significant differences between observed and expected species assemblages in this ecoregion at all urbanization levels and across all stream orders. EP assemblages showed less congruence among stream orders across urbanization levels. Urbanization was apparently more intense in 2nd-order sites than 1st- or 3rd-order sites; effects were potentially enhanced by the greater expected species richness (10) than in 1st- or 3rd-order sites (4 and 8, respectively; Table 7). The 1st- and 3rd-order sites ostensibly lost 1 and 3 species, respectively, of the expected assemblage at the >75% level of urbanization, whereas 2nd-order sites lost 5 species with this level of urbanization.

The success of one tolerant fish species (R. atratulus) was responsible, however, for maintaining a high abundance of fish in 1st-order urbanized EP sites, whereas in 2nd and 3rd order sites several intolerant species contributed to abundance as urbanization increased (see intolerant species list in Table 2). However, 2nd- and 3rd-order sites appeared more resistant to increasing urbanization than 1st-order sites, possibly because of increasing habitat size and species complexity, with abundance not dominated by any one tolerant species (Table 2). These results suggest that fish assemblages in the smallest streams are sensitive to urbanization, where fish abundances may be more variable than expected. With increasing habitat size and size of the fish species pool, assemblages in larger streams (2nd- and 3rd-order streams in our study) may be resistant to low levels of catchment urbanization (10-25%), but eventually become altered at higher urbanization (>25%).

One reason for low correspondence between assemblages and urbanization in CP streams is that species shifts may have already occurred in most streams within this ecoregion, irrespective of contemporary urbanization. This result was surprising because we expected to find dramatic differences in richness and abundance between most and least-urbanized sites (Tables 3, 6). This disparity could have resulted from changes in habitat and/or foodweb structure, invasion by opportunistic species, or a combination of these factors. Trebitz et al. (2003) warned that differences in life-history traits among species and the associated interdependence of component metrics within IBIs may reduce the IBI's utility in bioassessment; thus, such multimetric indices should be used cautiously when evaluating potential changes in species assemblages. In our study, the poor correspondence between FIBI and catchment urbanization in CP sites suggests that it may be more useful to base assessments on individual species traits and/or FIBI component metrics to elucidate assemblage–urbanization patterns.

Fish assemblages respond to many environmental factors, including spatial and temporal variation in interspecific interactions and stream hydrology (Lyons 1996, Paller 1994, Schlosser 1982, Angermeier and Winston 1999, Gorman and Karr 1978, Rahel and Hubert 1991, Grossman et al. 1998, Hughes et al. 1998). In particular, altered flow regimes from urbanization can affect fish assemblage structure and biodiversity (Bunn and Arthington 2002, Poff and Allan 1995, Roy et al. 2005). Flow shapes stream physical habitat, with concomitant influences on biotic composition; yet, fish populations often have evolved life histories that reflect natural flow regimes (Bunn and Arthington 2002). Rapid alterations in flow regimes in urbanizing streams, which may be the case in Maryland streams (CWP 2003), may have occurred on too short a time scale (years to decades) to allow populations to respond, thus exacerbating the urban syndrome (Booth 2005, Groffman et al. 2003,).

We examined correspondence between urbanization and fish assemblages at a broad (ecoregional) spatial scale; however, it may be more accurate to address such relationships at the smaller reach scale because assemblages may be more influenced by reach-scale conditions or processes (Wang et al. 2003). For example, changes in riparian conditions attributable to urbanization may alter channel complexity, which, in turn, may alter fish assemblages (Booth 2005). Our future research will assess which and how reach-scale habitat variables change with urbanization, which fish populations are most responsive to changes, and how catchment imperviousness (especially effective imperviousness, sensu Walsh et al. 2005a) within Maryland may be a stressor to fish assemblages. Such studies have important consequences to the entire Chesapeake Bay Catchment because of increasing development throughout the region.

Fish biodiversity in urban streams

The maintenance of fish diversity across Maryland also is an important consideration in understanding the consequences of urbanization (Roth et al. 1999) because many species classified as rare within the state occur in areas either affected by urbanization now, or will be in the future. Ricciardi and Rasmussen (1999) noted that human population growth is a major factor related to fish species extinction, especially in urbanizing areas. Unfortunately, conservation practices minimizing impact of urbanization on local or regional fish assemblages, especially in the Chesapeake Bay Catchment, may be inadequate, too late, or too expensive to protect intolerant fishes because of the invasiveness and nonreversibility of urbanization. For example, it will be logistically difficult, if not politically impossible, to reverse road density and catchment imperviousness within urban Maryland and throughout the US (Brabec at al. 2002). Wang et al. (2001) and Wang and Kanehl (2003) both suggested that minimizing connected imperviousness, or eliminating restricting catchment imperviousness (especially to <10–15% in a catchment) from the protecting riparian habitat, may be critical to maintaining species assemblages (Gergel et al. 2002, Groffman et al. 2003); we believe this recommendation also may be useful in protecting Maryland stream fishes.

Loss of fish refugia (needed to maintain biodiversity) within streams in urbanizing catchments is an environmental concern within Maryland (Richter et al.

1997). Maintenance of source populations and dispersal should be key considerations in urban planning efforts (Lowe 2002). Connectivity within catchments is being destroyed by urbanization, along with daily destruction of small perennial and intermittent sites (CWP 2003). Angermeier and Winston (1999) urged protection of fish biodiversity and species assemblages through enhanced protection of key processes at the landscape scale. Sites are the lifelines of the landscape and also integrate catchment processes; thus, their protection and restoration, especially in urban areas, are critical to maintaining economic vitality and providing ecological services.

Tables

Table 1. Distribution of 544 Maryland Biological Stream Survey (MBSS) sample sites within the eastern Piedmont and western shore Coastal Plain ecoregions of Maryland, USA.

Ecoregion	% of catchment urbanized	Stream Order		
		1	2	3
Eastern Piedmont	<10	83	39	35
	10–25	6	5	14
	25–50	23	6	12
	50–75	21	4	3
	>75	5	5	4
	n	138	59	68
Coastal Plain	<10	73	32	28
	10–25	29	22	7
	25–50	18	10	22
	50–75	13	9	4
	>75	8	4	0
	n	141	77	61

Table 2. Relative abundance (= abundance) and % of sites containing a given species (= constancy) of probable fish assemblages (Kilian 2004) collected from eastern Piedmont and Coastal Plain streams of Maryland, USA. T = tolerant species, I = intolerant species, U = tolerance unknown. Blank spaces = species did not occur.

	Stream order					
	1		2		3	
Species (Tolerance)	Abundance	Constancy	Abundance	Constancy	Abundance	Constancy
Eastern Piedmont						
Rhinichthys atratulus (T)	36.2	88.9	6.1	69.1	10.8	80.6
Semotilus atromaculatus (T)	27.1	76.7	8.2	90.9	8.1	81.3
Clinostomus funduloides (I)	17.2	51.1	17.2	98.2	4.0	59.0
Catostomus commersoni (T)	5.5	52.2	8.2	94.6	11.4	91.4
Etheostoma olmstedi (T)			6.6	94.6	5.8	61.9
Luxilus cornutus (I)			7.0	67.3	5.3	55.4
Rhinichthys cataractae (I)			10.6	72.7	9.7	80.6
Exoglossum maxillingua (I)			6.9	67.3		

Hypentelium nigricans (I)			2.0	54.6		
Anguilla rostrata (U)			6.1	69.1		
Lepomis auritus (I)					4.0	59.0
Lepomis macrochirus (T)					3.7	58.3
Coastal Plain						
Umbra pygmaea (T)	21.6	95.2	47.1	100.0	9.6	93.9
Anguilla rostrata (U)	13.4	90.5	5.7	48.3	20.3	98.5
Esox americanus (T)	9.7	59.5	3.4	78.3	3.5	71.2
Erimyzon oblongus (U)	7.4	57.1	9.9	73.9	6.0	68.2
Lepomis gibbosus (T)	5.5	69.1	3.8	65.2	4.1	89.4
Etheostoma olmstedi (T)	15.6	52.4	5.0	52.2	16.8	98.5
Aphredoderus sayanus (T)	11.6	54.8	7.7	60.9	5.3	68.2
Lepomis macrochirus (T)	13.7	69.1	4.7	52.2	5.2	83.3
Noturus gyrinus (U)			5.7	56.5	3.8	59.1

Notemigonus crysoleucas (T)	13.3	60.9		
Semotilus corporalis (I)			7.8	57.6
Notorus insignis (I)			2.2	59.1
Esox niger (U)			2.0	59.1
Micropterus salmoides (T)			1.4	56.1
Lepomis auritus (I)			6.7	66.7
Lamptera aepyptera (U)			4.7	62.1
Enneacanthus gloriosus (U)			3.5	57.6

Table 3. Mean (±1 SD) fish species richness and abundance in 1st-, 2nd-, and 3rd-order Coastal Plain streams with contrasting catchment urbanization. There were no significant differences in richness or abundance among urban categories for a given stream order (*n* = 279).

Stream order	% of catchment urbanized	Richness (no. species/site)	Abundance (no. individuals/site)
1	0–10	5.8 (4)	169 (168)
	10–25	4.7 (3.7)	141 (175)
	25–50	4.4 (3.8)	108 (117)
	>50	4.3 (4.6)	133 (109)
2	0–10	11 (5.1)	242 (204)
	10–25	12 (5.6)	220 (171)
	25–50	12 (6.3)	394 (187)
	>50	11 (5.1)	337 (272)
3	0–25	16 (5.1)	416 (512)
	>25	15 (4.6)	542 (1079)

Table 4. Mean (±1 SD) fish species richness and abundance in 1st-, 2nd-, and 3rd-order eastern Piedmont streams with contrasting catchment urbanization. Least Significant Difference (LSD) groupings for richness or abundance values for a given stream order with the same letter were not significantly different at $\alpha = 0.05$ ($n = 265$).

Stream order	% of catchment urbanized	Richness (no. species/site)	LSD	Abundance (no. individuals/site)	LSD
1	0–25	6.9 (3.9)	A	346 (339)	A
	25–50	4.5 (3.3)	B	162 (161)	B
	>50	2.8 (2.4)	C	139 (187)	B
2	0–25	12 (4.5)	A	560 (388)	A
	25–50	8.0 (2.3)	A, C	322 (254)	A
	>50	4.9 (2.3)	B, C	434 (445)	A
3	0–25	17 (4.4)	A	657 (371)	A
	25–50	13 (2.9)	B	365 (181)	B
	>50	5.1 (1.4)	C	201 (307)	C

Table 5. ANOVA results for fish species richness (no. species/site) and abundance (no. individuals/site) in 1st-, 2nd-, and 3rd-order Coastal Plain sites and eastern Piedmont sites. Coastal Plain (A) sites were grouped by 10, 10–25, 25–50, and >50% of catchment urbanized (1st- and 2nd-order sites), and 0–25 and >25% of catchment urbanized for 3rd-order sites, and eastern Piedmont sites (B) were grouped by 0–25, 25–50, and >50% of catchment urbanized.

Ecoregion	Site order	Metric	MS effect	MS error	*F* (df)	*P*
A: Coastal Plain	1	Richness	20.9	16.2	1.3 (3,137)	0.28
		Abundance	22,370	28,042	0.80 (3,137)	0.50
	2	Richness	4.2	29.1	0.15 (3,73)	0.93
		Abundance	97,706	42,477	2.3 (3,73)	0.084
	3	Richness	7.4	24.2	0.31 (1,59)	0.58
		Abundance	237,599	644,253	0.37 (1,59)	0.55
B: Eastern Piedmont	1	Richness	1.35	0.082	16.6 (2,135)	<0.000001
		Abundance	609,188	85,854	7.1 (2,135)	0.0012
	2	Richness	0.47	0.55	8.5 (2,56)	0.00060
		Abundance	186,141	149,603	1.24 (2,56)	0.30
	3	Richness	0.66	0.012	57.5 (2,65)	<0.000001
		Abundance	1.7	0.092	18.9 (2,65)	<0.000001

Table 6. Results comparing probable expected (bolded) to observed fish assemblages in Coastal Plain streams, based on % of catchment urbanized. Sites in "x" indicate nonsignificant differences ($P > 0.05$) between observed and expected assemblages for a given category of % catchment urbanization. Expected species richness was artificially lowered (italicized numbers) to determine when observed assemblages would meet model expectations (i.e., no difference between observed and expected richness at any urbanization level). Categories of % urbanization as in Table 1.

		% of catchment urbanized				
Stream order	Richness	0–10%	10–25%	25–50%	50–75%	>75%
1	**8**					
	7					
	6					
	5					
	4	X				
	3	X	X	X		
	2	X	X	X	X	X
2	**10**					
	9					
	8	X				
	7	X	X	X	X	
	6	X	X	X	X	X
3	**16**					
	15					
	14					
	13					
	12	X	X	X		
	11	X	X	X		
	10	X	X	X		
	9	X	X	X	X	X

Table 7. Results comparing probable expected (bolded) to observed fish assemblages in eastern Piedmont sites, based on % of catchment urbanized. Sites in "x" indicate nonsignificant differences ($P > 0.05$) between observed and expected assemblages for a given category of % catchment urbanization. Expected species richness was artificially lowered (italicized numbers) to determine when observed assemblages would meet model expectations (i.e., no difference between observed and expected richness at any urbanization level). Categories of % urbanization as in Table 1.

		% of catchment urbanized				
Stream order	Richness	0–10%	10–25%	25–50%	50–75%	>75%
1	**4**	X	X	X		
	3	X	X	X	X	X
2	**10**	X	X			
	9	X	X	X		
	8	X	X	X		
	7	X	X	X		
	6	X	X	X		
	5	X	X	X	X	X
3	**8**	X	X	X		
	7	X	X	X		
	6	X	X	X	X	
	5	X	X	X	X	X

Figures

Figure 1. Ecoregions and major catchments within Maryland, USA. The Piedmont Plateau Province (EP) consists of Lowland (west) and Upland (east) sections, whereas the Coastal Plain Province (CP) consists of the Western Shore Uplands (west, in part), the Western Shore Lowlands (west, in part), and the Delmarva Peninsula regions (east). We focused on the Western Shore Uplands and Lowlands regions of the CP, and the Upland Section of the EP. (Reprinted from Pirhalla 2004, with permission from the American Fisheries Society, Bethesda, Maryland).

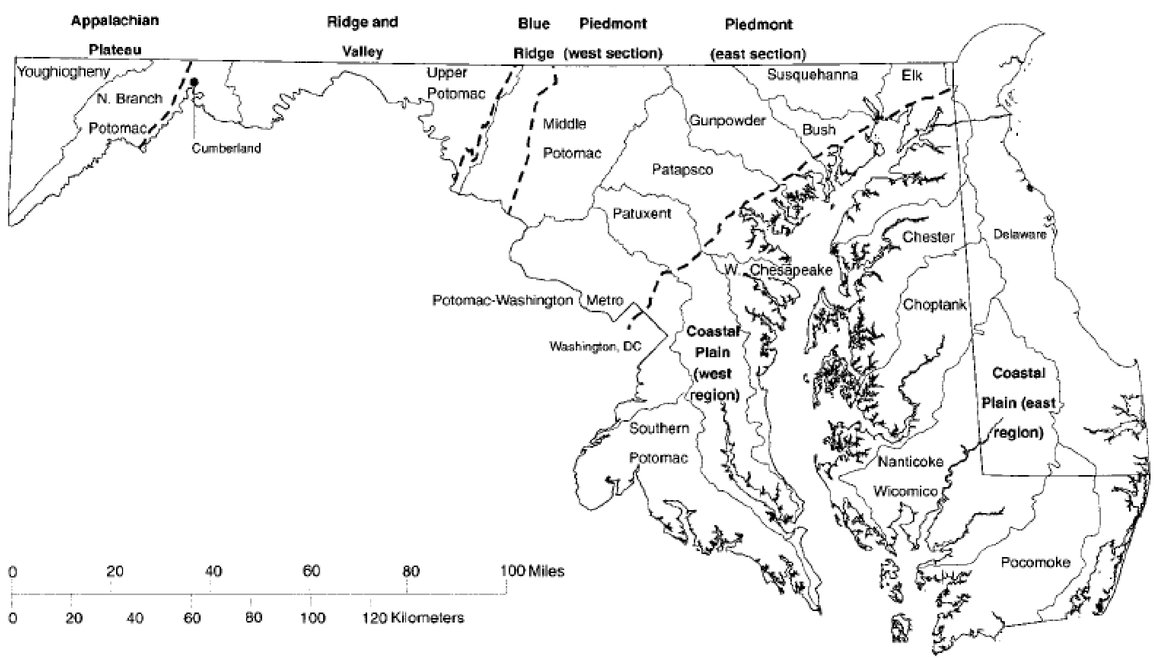
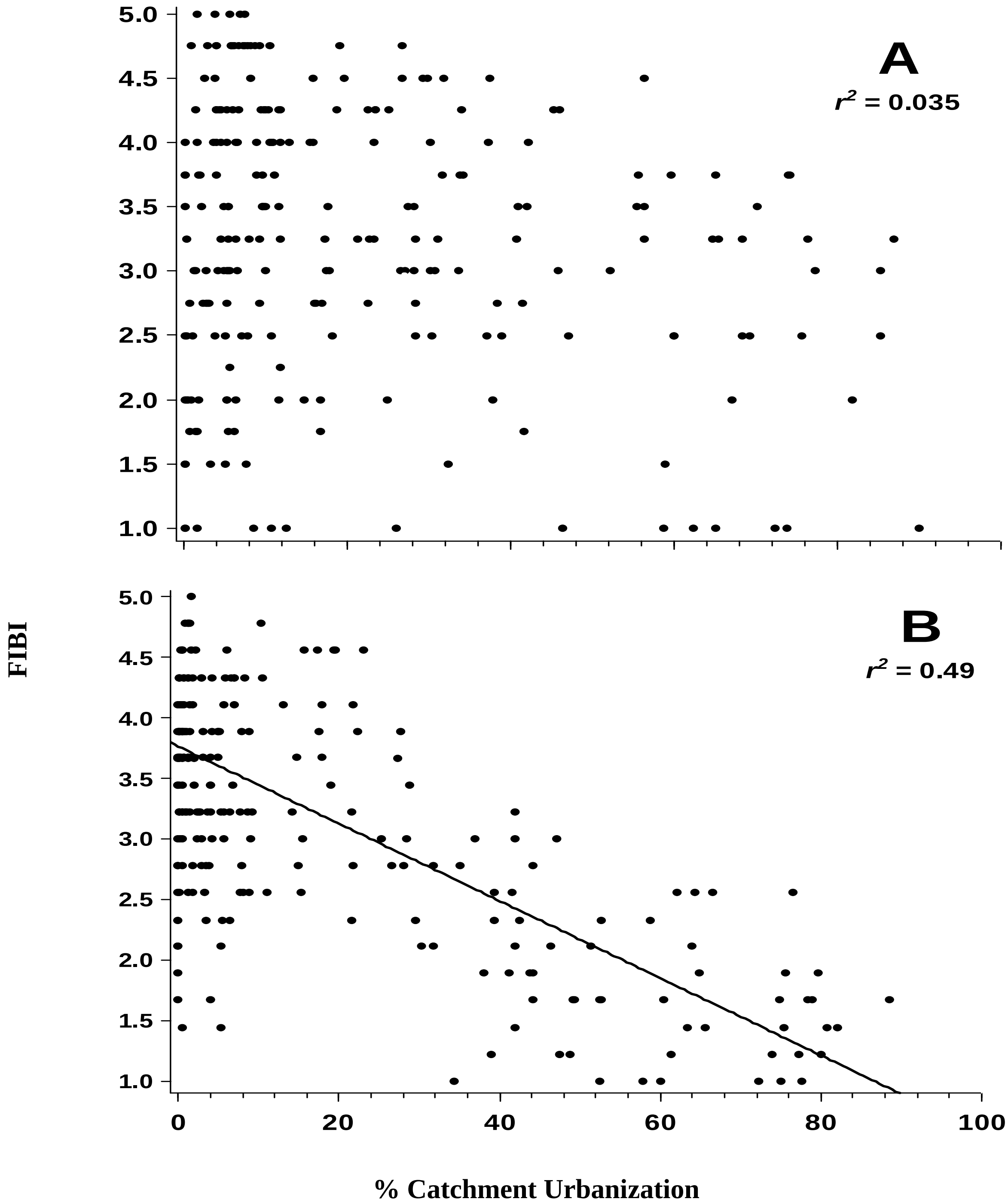


Figure 2. Relationship between % of catchment urbanization and the Maryland fish index of biotic integrity (FIBI) for 1st -, 2nd-, and 3rd-order Coastal Plain sites (A) and eastern Piedmont sites (B).



Chapter 3: Slip-sliding away: Changes in stream habitat complexity along the urban – rural land use gradient

Abstract

Stream habitat is shaped by the disturbance regime of water flow through the channel. As watershed composition changes along the urban – rural gradient, it is hypothesized that physical habitat attributes that are important for aquatic biota degrades due to the increased connectedness between urban land use and the stream channel. Specifically, changes in 1) the pattern of channel subunits, such as riffles, runs, pools, and glides; 2) the extent of erosion and bar substrate size; 3) water quality; and 4) the quantity of good instream habitat occur over this land use gradient as defined by the percent urban land use (ULU) in the upstream watershed. A habitat survey was conducted at 56 first-order stream sites in the eastern Piedmont of Maryland which incorporated features of channel formations, instream habitat, water quality, discharge, and riparian vegetation. Significant changes in stream habitat due to urbanization were found in streams with >30% ULU. Specific conductivity was higher in all streams with >30% ULU, and maximum height of erosion and number of dewatered woody debris was highest in streams with 45-60% ULU. The most urbanized streams had a considerable presence of engineered banks and longest bar formation. Although no differences occurred in the extent and number of channel subunits, urbanization does appear to effect aspects of erosion and bar formation, water quality, and instream habitat along the urban – rural gradient. Thus, altered stream complexity may play an important role in homogenization of stream biota in urban ecosystems.

Introduction

Stream environments are a patchy, heterogeneous mixture of channel habitat subunits, such as pools, riffles, runs, glides, and backwater, which exhibit a diversity of depths, water velocities, refuge types, and food sources (Allan 1995, Lake 2000). In any stream ecosystem (natural or anthropogenic), physical habitat is primarily driven by flow, which in turn establishes the biotic community (Bunn and Arthington 2002, Poff et al. 1997). Channel formation, habitat complexity and the degree of habitat patch disturbance varies spatially and temporally due to discharge profiles, local geology and topography (Bunn and Arthington 2002, Frissel et al. 1986). Increased disturbance in a system, including climatic extremes through floods or droughts, modifies habitat availability and patchiness, and can generate a new patch configuration for biota to inhabit (Lake 2000).

Anthropogenically-influenced stream ecosystems experience 'floods' of a different magnitude than rural streams due to the nature of the upstream watershed land use and subsequent altered hydrologic cycle (Paul and Meyer 2001, Poff et al. 1997). Land use change across the eastern United States has transformed the lands' surface through the cutting of forests, plowing of fields, and paving over of porous soils (Allan 2004, Griffith et al. 2003). Each of these land use practices has modified the quality and quantity of water that reaches stream networks in different ways. Urbanization, in particular, increases the proportion of precipitation that is routed directly to the stream channel (increased connectivity) instead of its natural hydrologic route through groundwater to river systems (Arnold and Gibbons 1996, Paul and Meyer 2001, Walsh et al. 2005a). The installation of stormwater drains, which prevent pooling on paved roads and parking lots, can create raging rivers in the smallest stream channels during a precipitation event. Poor water quality and increased stormflow discharge have limited, and in some cases, devastated the available habitat for many aquatic species that are intolerant of pollutants, sediment, and altered temperature and flow regimes (Wood and Armitage 1997, Shields et al. 1994, Walters et al. 2003). These stormflow environments have been hypothesized to be the cause of decreased species richness and abundance of fish (Morgan and Cushman 2005, Tabit and Johnson 2002, Roy et al. 2005), herpetofauna, and algae (Potapova et al. 2005), leading to homogenization of the overall biotic community (McKinney 2006, Scott 2006, Marchetti et al. 2006).

Habitat surveys are commonly performed when stream fish and other fauna are being studied in order to relate niche characteristics and preferences (Aadland 1993, Gorman and Karr 1978, Wright and Li 2002, Richards et al. 1996, Wang et al. 2003). Many studies use multivariate analyses to determine if there is a correlation between faunal presence, abundance, and density with the habitat qualities examined (Poff and Allan 1995, Wright and Li 2002, Richards et al. 1996). Most studies that associate urbanization impacts to changes in fish assemblages relate how physical habitat is modified within the stream channel (Roy et al. 2005, Wang et al. 2003), yet I am not aware of any studies focusing on changes that occur within the stream channel over the urban – rural gradient.

Homogenization of biotic communities implies a simplification and decrease in the diversity and richness of species present in a system (McKinney 2006, Scott 2006, Rahel 2000). However, few studies have documented homogenization of physical habitat in urban stream channels (Booth 2005). Ecological and habitat degradation can be severe after new construction, or decades after a watershed is developed. Therefore, spatial and temporal variability in channel response plays a major role in watersheds depending on the speed and scale to which land is developed (Wood and Armitage 1997, Strayer et al. 2003). Dominant urbanization impacts have been primarily related to altered flow regimes and associated effects, which have a direct relationship with the appearance and functionality of the stream channel (Paul and Meyer 2001, Walsh et al. 2005b). Rural streams that have little stress due to urbanization may have greater habitat complexity throughout the stream channel, demonstrated by the presence of small debris jams and instream woody debris that provide variability in water velocity and sources of refuge and food. Urban systems are thought to lack these habitat components, resulting in stressed biotic communities (Booth 2005, Walsh et al. 2005b).

In this study, I chose to specifically examine stream networks across a gradient of increasing urban land use to establish whether stream channel habitat quality changes as a function of urbanization, and if so, where these changes occur. Within this objective, I specifically hypothesized that increased drainage connectivity between urban land use and the channel changes the: 1) channel morphology and subunits characteristics; 2) extent of erosion and bar formation and substrate size; 3) water quality and; 4) the quantity of good instream habitat occur across the urban – rural gradient. I predict that stream complexity and heterogeneity of fish habitat structure are reduced in urban streams compared to rural stream networks in forested watersheds. By modeling physical habitat characteristics that vary in condition with the percent of urban land use within the watershed, a better understanding of urban impacts on stream ecosystems can be prioritized for future management purposes.

Methods

Habitat surveys were conducted from June to August 2004 and June to September 2005. Each habitat survey initially followed the protocol of the Maryland Biological Stream Survey (MBSS), as established by the Maryland Department of Natural Resources (MDDNR) physical habitat survey, but was further modified to meet the conditions and objectives of this study. Kazyak (2000) provides a complete list of the parameters measured and information on the MBSS.

Stream sites were selected from the Maryland Urban Fish (MUF) database created from the MBSS 1995-1997 (Round 1) and 2000-2004 (Round 2) datasets. This database represents randomly chosen stream sites that were previously sampled by the MDDNR or University of Maryland. Inclusion into this database was based on the following criteria. Each site included was required to have a complete and comprehensive data record for a 75 m stream segment that included physical habitat parameters, water quality, fish collection, and land use characterization within the watershed. Then, a set of environmental criteria was imposed on the datasets to exclude sites that may present biases or impacts other than urbanization on stream biota. The resulting MUF database included first, second, and third order streams in the eastern Piedmont (EP) and Coastal Plain (CP) physiographic provinces in Maryland with less than 65% agricultural land use in the upstream watershed and less than 8 mg/L dissolved organic carbon (e.g. not blackwater).

Multivariate statistical techniques were performed on EP first order streams from the MUF database to explore which parameters previously measured could give insight to stream characteristics that require further description in degraded stream systems. All variables related to channel habitat found in this database were used to determine if there were specific qualities explaining a majority of the variance in stream habitat change over the urban to rural gradient. Principal components analysis was used to reduce the dimensionality of the data and identify important variables. To be considered important, those components with an eigenvalue ≥ 1 and variables with eigenvector weightings > 0.30 were investigated further by incorporation into the new habitat survey conducted for this study.

Multivariate analysis of 12 parameters (habitat and land use characteristics) in the MUF database revealed 4 important principal components explaining 75% of the total variance ($n = 138$ sites). Of these, 2 components demonstrated a high correlation with urban land use and impervious surface (Appendix I). The combination of variables in the first principal component indicated that more transect measurements should be made across the stream to better understand channel morphology. I also chose to collect data on the number, bank location, and size dimensions of the rootwads and woody debris to better understand the relationship of variables in the second component.

Study sites

Within the MUF database, stream sites were selected for this study based on the presence of two fish species that were required for other studies in this research as well as their location within the EP ecoregion. Ecoregions are defined by spatial patterns and composition of geology, physiography, vegetation, climate, soils, land use, wildlife, and hydrology (USEPA 2006). In this case, the EP physiographic province of Maryland and the EP ecoregion (as defined by USEPA) overlap and all stream sites were located within these areas. An additional criterion of watersheds greater than 202 ha (500 acres) was originally set to select sites from the MUF database due to the fact that blacknose dace and creek chub are more often found in streams of this size. However, this condition was lifted due to the need for more sites in distinct urban categories. These sites were specifically chosen based on their % ULU and location within river basins already included in the study. Selected sites were categorized by the percent urban land use (ULU) in the upstream watershed (0-15, 15-30, 30-45, 45-60, and >60%) and the number of sites within each category was relatively consistent.

The 56 first order stream sites surveyed for this study were found in 7 counties between Baltimore and Washington, D.C. (Baltimore – 22, Baltimore City – 1, Carroll – 1, Harford – 7, Howard – 13, Montgomery – 11, and Prince George's – 1; Figure 1). A total of 25 sites were sampled during the 2004 sampling season, and 31 were sampled during the 2005 season, all of which were located in the EP in various river basins (Table 1). The total number of sites per urban category was 19 [0-15%], 8 [15-30%], 8 [30-45%], 8 [45-60%], 13 [>60%]. However, when the number of sites in each urban land use category were split by year, there were 10[0-15%], 2 [15-30%], 3 [30-45%], 2 [45-60%], 8 [>60%] surveyed in 2004 and 9 [0-15%], 6 [15-30%], 5[30-45%], 6[45-60%], and 5[>60%] surveyed in 2005. Although the original watershed size criterion (>202 ha) was lifted, the average watershed area for sites sampled over two summers was 310 ± 32 ha with a range of 29-2091 ha (Table 1).

Field measurements

I visited every stream site prior to doing a stream habitat survey. Sites were located using information from MBSS datasheets, road maps, and GPS. Once the approximate position of the MBSS sample segment was determined, a random 75m long segment was measured (based on the stream thalweg) and marked. My sampling segment was close but did not always overlap the original segment used by the MBSS.

The date, time, weather, crew and GIS coordinates were recorded while the 75 m sampling segment was marked. Flags were placed at equal increments along the stream bank (0, 15, 30, 45, 60, and 75 m) for easier assessment of distance between points. Field observations of the surrounding landuse, presence and sightings of fish, herpetofauna, plants, benthic macroinvertebrates, and birds were noted. The presence of exotic plant species was also recorded for each site.

A stream map was drawn for the entire sampling segment, including the relative position and lengths of channel subunits (pool, riffle, run, and glide), sinuosity, position of woody debris, rootwads, debris jams, tributaries, and bar formation. Estimated lengths of each channel unit were recorded separately, as well as the number of distinct units within the sampling segment. Along each streamwalk, the diameter (nearest tenth of a meter) of every rootwad was measured, and each tree species was identified to the lowest level possible. A tree was considered a rootwad if it was still alive and maintaining streambank stability with at least some roots exposed and considered woody debris if dead and found either in the stream or within 5 m of the streambank edge. Woody debris were measured for length (m) and approximate diameter (nearest cm), but only those ≥ 10 cm in diameter and 1 m in length were recorded. The number of woody debris and rootwads were tallied for both left and right bank separately. Similarly, the number of debris jams (wedged piles of woody debris and other organic matter greater than 0.25 m^2) were tallied for left and right banks, as well as those found in the middle of the stream channel. If tributaries were present within the sampling segment, the position and width along the main sampling segment was recorded. The presence of foot-bridges and roads across the stream within sight from the segment was also tallied.

The linear extent of visibly eroded streambanks as well as the maximum height of erosion was estimated and recorded for each 75 m segment. This was done by visual estimates within each 15 m sampling increment. To complement the stream channel unit characterization, measurements of bar formation were also performed. The linear length, side of the channel, position within the 75 m sampling segment, and the sediment composition (based on size) were recorded. Presence of any vegetation or other stabilizing cover found on these bars was noted. Sediment types were characterized by size, following sediment standard class sizes - silt (<0.1 mm), sand (0.1 – 1 mm), gravel (1 mm – 25 cm), cobble (25 – 100 cm), boulder (>100 cm), and bedrock.

Water quality measurements were made above the 75 m sampling segment so as to not sample in disturbed water. A Hydrolab® Quanta® was placed in the middle of the stream channel to collect single recordings of stream water temperature (°C), pH, dissolved oxygen (mg/L), and conductivity (mS/cm) measurements. Depth (m) and velocity (m/s) were measured at regular intervals across a constrained width of the stream (m) to estimate discharge (m^3/s).

In addition, stream channel transect measurements were made at each flag (15 m intervals) to give a more comprehensive picture of the study site. Channel characteristics including stream width, thalweg depth, and thalweg velocity were recorded. The percent shading over the channel, as well as the type and extent of cover within a 10 m buffer of riparian vegetation, was described.

Analyses

Many of the variables that were measured separately between left and right banks were summed for analysis. This included the number of instream rootwads, dewatered rootwads, instream woody debris, dewatered woody debris, debris jams, pipes, as well as the linear extent of erosion, undercut banks, bank stabilization, bar formation, and gabion. The maximum height of erosion for left and right banks was averaged for each stream. Among many of the measured parameters, I calculated the total amount of engineered banks by summing the linear extent of gabion and other bank stabilization techniques. The width:depth ratio was also calculated by dividing the average width by average depth measurements. The average width, depth, and shading over the channel was calculated using the six transect measurements for each stream. Finally, the area of a rootwad was calculated using the measured diameter of the exposed rootwad and the equation for area of a circle. Woody debris surface area was estimated using the average diameter and length of the log in the equation for surface area of a cylinder. Likewise, volume was calculated using the equation for the volume of a cylinder.

Analysis of variance (ANOVA) was performed on many of the variables representing habitat complexity to determine if differences occurred across the urban gradient. Variables included in this analysis were temperature, dissolved oxygen, pH, conductivity, extent of pool, glide, run, and riffle, erosion, maximum height of erosion, instream and dewatered rootwads and woody debris, undercut banks, bank stabilization, debris jams, bar formations, tributaries, pipes, gabion, engineered banks, bridges, discharge, average width, depth, and shading, and width:depth ratio. This two-way ANOVA tested the effects of both urban category and year sampled, utilizing a least squares means test to isolate specific differences between years and urban categories. I also tested the data to see if differences occurred between dewatered and instream rootwad area as well as woody debris volume and surface area across the urban land use gradient. Differences were considered significant at $P < 0.05$ using SAS statistical software (SAS Institute 1999).

I conducted a stepwise multiple regression analysis to determine whether stream habitat degradation was explained by complex, multivariate relationships. Multiple regression has been used in many studies when there are multiple possible predictor variables and one response variable (Tong 2001, Tong 2003, Holland et al. 2004) Stepwise multiple regression scans all possible variables given, choosing those that provide the greatest explanation of the response variance in decreasing order (Gotelli and Ellison 2004). This analysis allows variables to enter and leave the regression equation depending on how high an R-squared (R^2) the variable combination generates. Although SAS uses a default setting of $P = 0.15$, I selected 0.05 for forward and backward entry into the equation. The multiple regression equation, R^2, and Mallow's C (C_p) are reported for variables which implied that either %ULU or % impervious surface were important to the relationship. Mallow's C is a diagnostic tool that indicates how well the model describes the tested relationship. Low values of C_p relay the best model selection. The predictor variables were also checked for collinearity using variance inflation factor (VIF) analysis, where VIF values less than 10 reflect a lack of collinearity (Belsley et al. 1980). Finally, I performed a simple linear regression on each of the resulting response variables against the significant urban land use variable, reporting the linear equation and adjusted R^2.

Results

Analysis of variance

Analysis of variance indicated that urban land use explained differences in nine variables (Table 2). Three of these variables (erosion, pipe, and discharge – Ucat "effect") did not generate any specific differences after adjustment for post hoc comparisons. However, conductivity was significantly different across urban categories (F = 6.2; df = 4, 46; $P < 0.001$; Table 2). Watersheds with 0-15% ULU had lower stream conductivity than those with 30-45% (t = -3.3; df = 46; $P < 0.05$), 45-60 % (t = -3.9; df = 46; $P < 0.01$), and > 60% ULU (t = -3.6; df = 46; $P < 0.01$; Figure 2). Bank stabilization, including large cobble, boulders, fiber netting or other man-made structures, was significantly greater in streams with > 60% ULU than in those with 45-60% (t = -3.4; df = 46; $P < 0.001$), 30-45% (t = -3.1; df = 46; $P < 0.05$) or 0-15% ULU (t = -4.6; df = 46; $P < 0.01$; Table 2 and Figure 3). Although the extent of gabion (wire containers filled with stone) was not significantly different across the land use gradient, when gabion and other forms of bank stabilization were combined, urban streams exhibited many more linear meters of engineered banks (F = 5.8; df = 4, 46; $P < 0.001$; Table 2). Between urban categories, those streams with > 60% ULU had significantly more engineered banks than those with 0-15% (t = -3.3; df = 46; $P < 0.05$), 30-45% (t = -3.2; df = 46; $P < 0.05$), and 45-60% ULU (t = -4.5; df = 46; $P < 0.001$; Figure 4). Finally, bar formation was also greater in the most urbanized streams than in those with 0-15% (t = -2.9; df = 46; $P < 0.05$) 30-45% (t = -3.0; df = 46; $P < 0.05$), and 45-60% ULU (t = -2.5; df = 46; $P < 0.05$; Figure 5).

Exploration of channel subunit data did not reveal any significant differences in the extent of riffles, runs, pools, and glides among land use categories (Table 2). However, the amount of glide habitat generally increased and the extent of riffles decreased as the % ULU increased (Table 3). The extent of runs was highest in the most urban streams and lowest at 15-30% ULU sites (Table 3). Pool habitat was found in greatest abundance at 15-30% ULU sites, and in lowest abundance, surprisingly, in urban streams (Table 3).

Some of the data demonstrated a sampling year effect. There was a difference between years in erosion extent ($F = 2.8$; df = 4, 46; $P < 0.05$; Table 2) where streams sampled in 2005 had more eroded surfaces than those sampled in 2004 ($t = -2.6$; df = 46; $P < 0.05$). Other variables displayed significant differences between years as well. The average extent of riffles was 20.6 ± 2.76 m in 2004 and 16.7 ± 1.99 m in 2005 ($t = 2.3$; df = 46; $P < 0.05$). Glide extent was also greater in 2004 than in 2005 (13.1 ± 2.49 vs. 5.1 ± 2.38, respectively; $t = 2.0$; df = 46; $P < 0.05$). The only difference in instream rootwads was indicated between years, where sites surveyed in 2005 had greater average densities (4 ± 0.5) than those surveyed in 2004 (2 ± 0.4) across the urban gradient ($t = -3.03$, df = 46, $P < 0.01$). Similarly, 2005 stream sites had more instream woody debris (4 ± 0.6) than 2004 sites (2 ± 0.4; $t = -3.0$; df = 46; $P < 0.01$).

Examination of both the number of dewatered woody debris and the maximum height of erosion along streambanks suggested differences among urban categories for the two years combined, as well as land use differences within years. Over the entire study, the number of dewatered woody debris was highest in streams with 45-60% ULU, indicating a significant difference with both the 15-30% ($t = -3.1$; df = 46; $P < 0.05$) and

30-45% ULU categories (t = -2.9; df = 46; $P < 0.05$). When the data for each year was separated, these significant differences occurred only in 2004 sites (Figure 6). Streams with 45-60% ULU had on average 18 ± 0.5 dewatered woody debris compared to 2 ± 0.5 in 15-30% ULU and 1 ± 1.3 in 30-45% ULU streams (Figure 6).

There was an urban land use effect within the maximum height of erosion as well (F = 6.0; df = 4, 46; $P < 0.001$) suggesting a similar peak in streams with 45-60% ULU (Figure 7). These streams had significantly higher eroded banks than streams with 0-15% (t = -4.6; df = 46; $P < 0.001$), 30-45% (t = -3.7; df = 46; $P < 0.01$), and surprisingly > 60% ULU (t = 4.1; df = 46; $P < 0.01$). However, there was also a sampling year effect (F = 8.9; df = 4, 46; $P < 0.01$), revealing that erosion was higher in 2004 than in 2005. Streams within the 45-60% ULU category were significantly more eroded than all other categories ($P < 0.001$; Figure 7).

Analysis of rootwad area demonstrated that a significant difference occurred between rootwad types (F = 11.2; df = 1, 554; $P < 0.001$), although no land use effects (F = 1.4; df = 1, 554; $P = 0.22$) were present. Instream rootwads were on average, larger (13.4 ± 1.24 m) than dewatered rootwads (8.32 ± 0.62 m; t = -3.4; df = 554; $P < 0.001$). Instream rootwads at streams with 0-15% ULU were also larger than dewatered rootwads at streams with 0-15% ULU (t = -3.8; df = 554; $P < 0.01$), 30-45% ULU (t = -3.7; df = 554; $P < 0.01$) and 45-60% ULU (t = -3.3; df = 554; $P < 0.05$). When the estimated volume and surface area of woody debris (both instream and dewatered) were tested in the same analysis, there were no significant differences among either woody debris type or land use category.

Regression analyses

The stepwise multiple regression provided two significant relationships that complemented the ANOVA results. The variance in the linear extent of erosion was explained in 4 steps using $P = 0.05$ as entry and exit criteria in concurrence with % impervious surface, extent of riffles, dewatered rootwads and the extent of gabion along the streambank (Table 4). This multiple parameter equation explained 48% of the variance in erosion across the urban land use gradient ($C_p = -1.4$). In addition, conductivity was explained by 3 steps in coincidence with % impervious surface, pH and the extent of pools (Table 4). Forty-seven percent of the conductivity variation across the land use gradient was explained by these three variables ($C_p = 0.44$). None of the predictor variables found in either of these relationships were collinear.

A significant relationship between % urban land use and % impervious surface was confirmed by performing least squares regression on these parameters ($F = 325$; df = 1, 54; $P < 0.0001$; Adj.-$R^2 = 0.85$; Figure 8). This relationship suggests that each hectare of ULU is comprised of about 0.33 hectare impervious surface. As a result, I regressed both of the response variables found in the exploratory stepwise multiple regression analysis on % impervious surface to predict their relationship across the urban – rural gradient. Percent impervious surface predicted 12% of the variance in the linear extent of erosion along the streambank ($F = 8.64$; df = 1, 54; $P < 0.01$; Figure 9), and 26% of the variance in conductivity across the urban land use gradient ($F = 21.51$; df = 1, 54; $P < 0.0001$; Figure 10).

Observational data

Finally, a simple percentage analysis of observational data including fish, herpetofauna, benthic macroinvertebrates, exotic plants, substrate size classification of bar formations, as well as evidence of trash and sewer lines exhibited a few important trends. At only 4 of the 56 sites were fish not observed, 3 of which were in the 45-60% ULU category, while the last was in the > 60% ULU category. Herpetofauna were visibly absent at 17 sites, with the highest percentage (50%) of absence at sites with 15-30% ULU. In addition, I did not see any benthic macroinvertebrates at 12 of the 56 sites. Benthics were seen at all sites within the 0-15% and 30-45% ULU categories, but were not observed at 63% of the 15-30% ULU sites, 50% of the 45-60% ULU sites, as well as 23% of the >60% ULU sites.

Multiflora rose (*Rosa multiflora*) was the most commonly seen exotic plant species, present at 95% of all sites. Japanese stilt grass (*Microstegium vimineum*) and garlic mustard (*Alliaria petiolata*) were found at 57% and 34% of sites, respectively. In addition, mile-a-minute (*Polygonum perfoliatum*) was seen at 23% of sites, Japanese honeysuckle (*Lonicera japonica*) at 21% of sites, bull thistle (*Cirsium vulgare*) at 5%, and bamboo (*Bambusinae spp.*) at 2% of sites.

The substrate of bar formations was comprised mostly of sand, gravel and cobble across all stream sites surveyed (98, 96, 86% respectively). Silty bars were found at 32% of sites, while boulders were observed at only 20% of stream sites. Scoured bedrock was found at only 2 (5%) sites. Within the land use categories, streams with 45-60% ULU most frequently displayed silt, followed by the most rural streams (Table 5). Boulders were found in highest abundance in both 30-45% ULU streams as well as the most urbanized streams (Table 5). Finally, exposed bedrock was found only in streams with greater than 30% ULU (Table 5).

Evidence of trash was scored by any form of human constructed or fabricated materials, including paper and plastic trash, scrap metal, concrete, rubber, or appliances. Trash was observed at 39% of all stream sites, increasing over the land use gradient from 23 – 63%. Interestingly, sewer pipelines were also found at 20% of sites across the urban gradient, also revealing a relatively common presence across all urban categories (2: 0-15%, 2: 15-30%, 2: 30-45%, 1: 45-60%, 4: >60%).

<u>Discussion</u>

Changes due to urbanization

Although urbanization impacts are evident at very low % ULU, this research indicates significant changes in stream habitat when watersheds are composed of greater than 30% ULU. Water quality in streams with greater than 30% ULU displayed significantly higher conductivity than rural (0-15% ULU) streams. This could potentially be due to heavy road salt residuals in urban areas, since study on Baltimore streams across the land use gradient indicated a strong relationship between impervious surface and instream chloride levels even through summer months (Kushal et al. 2005). Evidence of increased conductivity has also been found in Australian (Hatt et al. 2004) and Georgia, USA (Rose 2002) urban streams. Streams in this category also exhibited the first presence of exposed bedrock and the highest density of boulders. Exposed bedrock is an indication of scour and runoff zones (Gomi et al. 2002) while the increased conductivity is most likely due to sediment transport from upstream bank erosion, direct runoff, or exposure to pipes (Paul and Meyer 2001).

Once watershed urbanization reaches the 45-60% ULU, stream habitat has degraded significantly. The greatest evidence of erosion (in height) and density of dewatered woody debris are found in these streams. In addition, the highest frequency of bars composed of silt (<0.1 mm) and the lowest frequency of cobble (25-100 cm) are present at streams with over 45% ULU. Trash was most frequent (63%) in streams with 45-60% ULU. In these urbanizing channels, dewatered woody debris represent logs that have fallen towards the stream channel due substantial undercutting from high storm flows. Study segments in two of these streams in the Patapsco River basin had 17 and 18 dewatered woody debris logs dispersed along the banks. Both of these sites were highly incised, with steep banks on either side, accounting for the maximum height of erosion, indicating major downcutting due to hillslope-stream interactions (Gomi et al. 2002, Shields et al. 1994). The high density of dewatered woody debris may provide good habitat once the logs fall into the water. Hilderbrand et al. (1997) found that LWD recruitment from riparian zones provides the best maintenance of channel elements and become important during bankfull discharge events. In urban settings, erosive undercutting of stream banks may increase recruitment of LWD to the stream channel. Because of these physical forces, the channel reacts by deposition of small sediment in the form of bars downstream from the site of erosion. The lower frequency of cobble-sized substrate in these bars and overall extent of bars may be due to nature of the soils and underlying parent material as well as the slope of the stream channel (Gomi et al. 2002).

Mature, urban stream channels provide the most comprehensive picture of erosional forces within a confined space. The extent of bar formation is at its peak in streams with >60% ULU, coincident with the high extent of erosion along stream banks. Not surprisingly, these streams also have the greatest extent of engineered banks, including loose, natural bank stabilization techniques, such as willow plantings with fiber netting, as well as wired gabion. This is most likely a result of restoration projects directed by local and state natural resource managers to reduce erosion and re-directing of the stream channel from high stormflows. Engineered stream banks may reduce the height of erosion, leading to a reduction of channel incision, however, the magnitude and power of urban stormflow produces a dynamic morphological setting that creates longer, more extensive bars. Thus, channel habitat quality decreases in these streams even though there is an increased presence of bank stabilization (Shields et al. 1994).

Highly urbanized channels are generally devoid of small sediment, confirmed by the lack of silt in bars, in contrast to a high frequency of silty bars in rural streams with less transport downstream. Pipes are most common in urban streams as well, but are also found in rural, less-impacted streams too. This is most likely due to the fact that rural streams sites were many times found within the close vicinity of a road, and drainpipes were counted within this tally. Sewerlines, which were not included in the pipe tally were also most frequent in mature urbanized stream networks. These large pipes with access towers may not have been originally established within the channel, however many of them were found either within, exposed to, or just outside of the wetted stream channel. Stream valleys provide a path of least resistance and easy access for sewer and clean water networks, so it is logical that many of these sites were concurrent with public water and sewer lines. Trash was also high at sites with >60% ULU, although not as high as in 45-60% ULU sites. However, this common source of bacteria, channel clogging, and poor aesthetic appeal can be due to a variety of sources. The fact that trash was found at even the most rural sites is indicative of a lack of control and/or respect for stream ecosystems. More than a few urban sites could have been characterized as public dump sites, spread with large pieces of metal, tires, shingles, even appliances (refrigerator and water heater to name a few!). In many of these cases, the abundance of trash was most likely due to the closest landowner, previous landowners, or the degradation of old buildings and bridge structures. At other sites, including many of the urban streams, it was most likely due to non-point sources, such as the accumulation of trash from city streets that was washed into the stream during the last precipitation event.

Unexpectedly, temperature did not reveal any changes across the urban gradient. This was most likely due to the fact that stream temperature was measured only once at each of the study sites, at times between 8 am and 6 pm, throughout the summer months. Individual measurements record a snapshot of time, which does not provide the temperature profile that may be required to understand differences between urban and rural watershed processes. This is quite interesting though, since temperature differences were observed in point data collected in a following study (Chapter 5) and have been documented in other research (Brasher 2003, Paul and Meyer 2001).

Additionally, I expected to see differences in the average channel subunit lengths across the urban – rural gradient. Although there were some interesting trends within the data, I predicted that the length of pool habitat would be greater in urban stream reaches due to the increased presence of pipes and culverts that scour and create longer pools.

Although the number of pipes increased in urban streams, the length of pool habitat did not. It was not surprising that run habitat was most abundant in urban streams. Two of the most abundant fish in urban streams, *Rhinichthys atratulus* blacknose dace and *Semotilus atromaculatus* creek chub, inhabit runs and pools (Chapters 4 and 5, Morgan and Cushman 2005, Jenkins and Burkehead 1993). Streams with greater than 45% ULU had a combined run-pool extent of ~ 52 m out of 75 m, in which I would expect an increase in the fish abundance. Since the amount of habitat is not significantly different in urban as compared to rural streams however, there must be other factors related to the abundance of these particular fish species.

Differences due to sampling year

The differences I found due to the year streams were sampled were unexpected. Many variables including the extent of riffles and glides, the maximum height of bank erosion and the number of dewatered woody debris were all higher in 2004 than 2005. Conversely, the linear extent of bank erosion and number of instream woody debris and rootwads were greater in 2005 than in 2004. The reason for these differences are unknown, however there are two potential explanations. First, there could be specific site or geographic distinctions in how the stream channels responds to upstream urbanization. For instance, two sites visited in 2004 in the 45-60% ULU that displayed major differences in the height of erosion and number of dewatered woody debris (Figures 6 and 7) were in very close geographic proximity to each other (about 3.2 km). These two streams (BA-117- 2004 and PATL-119-2004) are small headwater streams that eventually flow into the lower Patapsco River through adjacent tributaries. Three of the sites sampled in 2005 within the 45-60% urban category were also found within the

Patapsco basin, but were located further north in the Gwynns Falls and Loch Raven Reservoir subwatersheds. The remaining 3 streams within this category were found in the Gunpowder (2) and Bush river (1) basins, which are also geographically north of the 2004 sites. Thus, the degradation seen in the 2004 streams as compared to the 2005 streams could be due to geographic location and inherent topographic and geologic differences. Indeed, it could also be due to the fact that these streams experience different urban stressors as well, related to the geographic location or land use within the subwatersheds, even though the percent urban land use is similar to others sampled in 2005.

Secondly, this divergence in habitat variables across years could be due to climatic differences. The linear extent of erosion in 2005 as well as the number of rootwads and woody debris found in the stream may be due to greater precipitation over the course of the summer months. Instream habitat structures are defined as being partially or completely submerged below the waters' surface, while dewatered structures are found just above the channel or immediately adjacent to the wetted channel. If baseflow was higher at these streams due to steady rainfall throughout the summer 2005, it is likely that more rootwads and woody debris would be considered instream versus dewatered (at lower baseflow levels). The large amount of dewatered woody debris in 2005 could have also been due to a few large precipitation events in spring or summer, causing instream woody debris to be transported downstream to a resting place outside of the channel. Precipitation in the Baltimore, MD region was slightly higher in 2005 than in 2004 (41.19 vs. 39.59 cm) from June 1 to August 31, however the maximum single rainfall was much higher in 2004 (11.3 vs. 7.1 cm in 2005; Weather Underground History

2006). Storms causing major treefall are also a potential cause for higher density of dewatered woody debris (Gomi et al. 2002). Thus, seasonal and climatic effects are also a source of variation in year-year results. In conclusion, the basis for yearly differences in habitat variables is most likely a combination of both geographic and climatic variation.

In addition to site differences leading to yearly variation, there was another example of an outlier that produced some noteworthy disparities. LWIN-120-2005, found in the Bush River basin was grouped into the 45-60% urban category due to its 57.3% ULU, however its % impervious surface was 37.62 which is two times higher than many of the other sites in this category. In this case, the stream channel was adjacent to Interstate 95 in Harford County, which significantly increases the amount of impervious surface within the upstream watershed. Other sites, including LIGU-105-2005 and SENE-114-2005 were on the other end of the spectrum with respect to % ULU and % impervious surface. These sites had 31.71 and 13.11% ULU, respectively, however very little % impervious surface (0.09 and 0.11 %, respectively), but there is no apparent reason for this discrepancy. Site differences obviously lend increased variation to any relationship, and are important to discuss.

Land use legacies

Across the region of stream sites sampled, invasive plant species were found in a relatively consistent manner. This is a key indication of the past land use legacy of disturbance in Maryland. In the Piedmont physiographic region, land that was originally cleared for agriculture as well as forested land has been transformed into urban land cover at an increasing rate since the 1970's (Griffith et al. 2003). Since multiflora rose was present at 95% of stream sites, is a prolific species in open woodlands, forest edges and along streambanks that signals disturbance, it is reasonable to suggest that most of the stream sites surveyed in this study have been exposed to some type of disturbance (natural and/or anthropogenic; Multiflora Rose 2006). In addition to urbanization, mixed land use (of which most of the studied watersheds are) generates a suite of stressors that are difficult to tease out. Studies along the urban gradient encounter added "environmental" factors such as socioeconomic, population, and infrastructure complexity in determining the response of natural systems to a stress (McMahon and Cuffney 2000). In this analysis of habitat changes along the urban – rural gradient, urban land use was employed to predict changes in stream habitat which revealed a large amount of variation in response. This may be due to the fact that ecological processes do not always change linearly as the amount of anthropogenic impact increases (Theobald 2004). However, erosion and its associated impacts emerged as an important element in differentiating channel habitat (or lack thereof) among streams with increasing urban land use (Hammer 1972, Trimble 1997). In even the least urbanized systems, changes in channel habitat may be more accurately described if the proximity of roads to stream channels is accounted for, providing a direct linkage between sources of high stormflows and extent of erosion (Angermeier et al. 2004, Wheeler et al. 2005). Jones et al. (2000) concluded that road networks intensify floodwater energy resulting in debris flows and patches of disturbance within the channel as well as in the riparian zone downstream of road crossings. To make matters worse, there is a general lack of knowledge of the consequences of roads on aquatic biota (Angermeier et al. 2004). Even though it is known that impervious surfaces largely modify the channel morphology (Walsh et al.

2005a, Walsh et al. 2005b), the connection between instream habitat and biotic integrity is still relatively unknown.

In addition to aspects of spatial scale, there is a temporal scale that is important to consider within this urban framework. "Mature" urban systems, those that have been developed for decades, such as many of the urban sites surveyed in this study surrounding Baltimore City (~ 8 of 13 urban sites), exhibit different responses to stressors than developing urban systems. Similarly, watersheds that have been continuously developed over 5-10 years (in typical urban sprawl fashion) may exhibit less severe channel modifications than those that have been developed quickly within the last few years. Rapid changes in land use combined with extreme variability in precipitation can result in instant stream habitat degradation within a year due to erosion of construction sites. Furthermore, development of land that was previously farmed may produce different instream effects than land that was previously forested, potentially accounting for some of the variation found in this dataset. Thus, historic land use and the temporal scale over which land practices change are important to consider when assessing the impacts of land use on stream habitat.

Limitations

The small number of habitat components associated with urbanization from the multivariate procedures at the beginning of this study may have provided some limitations to interpretable results. My intent to identify important habitat variables that required further study stemmed from the concept that exploratory multivariate analyses can lead to scientifically plausible hypothesis testing within a large dataset with many variables. Since this analysis led to only two meaningful components to study further, I chose to generally expand on the habitat characteristic assessment that was originally created by MDDNR in the MBSS. Thus, my interpretation of stream channel habitat complexity and the variables that describe it may have neglected other important aspects of urbanization impacts that remain unforeseen.

Among previous studies of urbanization impacts, some use % ULU, while others have used % imperviousness. Significant changes in stream fish and other biota indicate the presence of thresholds around 10% imperviousness (Wang et al. 1997, 2001, 2003). Interestingly, the relationship between %ULU and % impervious surface in my study indicated that a watershed with ~33% ULU equals ~10% impervious surface, coinciding biotic effects in other studies to the first significant differences in this study of stream habitat. Although not all urban land use is created equally, we used %ULU to incorporate all aspects of urbanization, not just the imperviousness.

Some hypothesized differences were not detected in this study, which could be a result of the %ULU categories of urbanization used. The categorization of urban land use into increments of 15 was chosen based on results from Morgan and Cushman (2005), who used increments of 25 % ULU. Initially, I hypothesized that the data would suggest the presence of ecological thresholds similar to the aforementioned studies of stream biota. For this reason, I chose to use categories of % ULU to compare habitat quality across the urban – rural gradient. A threshold did occur in some of the parameters measured, specifically conductivity, maximum height of erosion, dewatered woody debris, and bar formation. Although conductivity indicated a threshold, the data suggests that it also increased linearly as the %ULU increased. In support of this, using % impervious surface, regression analysis of conductivity and extent of erosion suggested continuous functions. In this particular case, there is more evidence that conductivity increases as a continuous function across the urban – rural gradient. However, based on the other results from this research, some parameters do in fact present an ecological threshold. Thus, ecological thresholds and continuous relationships may both occur in different variables within the same ecosystem.

Finally, detecting change within a natural system can be difficult. I conducted a physical habitat survey along the urban – rural gradient to indicate where changes in stream habitat might occur. Other experimental approaches such as a paired watershed design of rural and urban streams might have indicated larger differences and more explicit relationships, however changes in habitat attributes that occur with intermediate levels of urbanization would be completely missed.

High variation in stream habitat to urbanization impacts was present in this study and therefore contributed to the lack of response in some important features for stream biota. Site selection, month of survey, and site location across a large metropolitan region may also add to potential previously discussed biases in this research. Each of the stream sites were selected from the MUF database, which was a subset of the MBSS database. The MBSS database contains streams sites that were randomly selected from the statewide stream network. Thus, although these sites represent a random set, the MUF database and more specifically, the sites chose for this study were not. Additionally, streams were surveyed throughout the summer months, which introduce great variability especially with respect to water quality and discharge. Temporal and spatial patterns of development within a watershed are important in determining changes within the stream channel and thus may also present a major source of error. Although these and other sources of error suggest that physical habitat does not change consistently across the land use gradient, natural variability was expected due to location of stream sites in five major river basins.

Summary

This study set out to test four hypotheses relating changes in stream channel morphology and habitat over the urban – rural gradient. The first hypothesis that changes occur in channel morphology and subunit extent was rejected. Although trends towards more glide and less riffle habitat were evident in urbanized streams, there were no significant changes in channel subunits along the urban – rural gradient. Secondly, while there was no significant increase in the number of storm drain pipes or other constructed drainages over the urban – rural gradient, the maximum height of erosion and linear extent of bar formation was significantly greater in urban systems. In addition, streams within urbanizing watersheds displayed bars commonly composed of silt, sand and gravel, yet heavily urban bars were composed of larger substrate sizes. The steady increase in stream water conductivity was a significant finding in this study, revealing a decline in water quality along the urban – rural gradient. Finally, although the number of instream woody debris and rootwads did not significantly decline (as one indication of good instream habitat), erosion played a large role in describing the changes that occur within the streambanks of urban systems. The considerable presence of engineered banks and other stabilization techniques convey past impacts of upstream urban land use within the stream. Although these structures serve their function well in most cases, they provide no ecosystem services to the aquatic or riparian biotic community (Shields et al. 1994). The transport of fine sediment associated with erosion, which is another serious stress to many aquatic biota, can clog interstitial spaces for invertebrates and create poor breeding habitat for fish (Wood and Armitage 1997). Thus, instream habitat quality declines along the urban – rural gradient to a point, after which heavily (> 60%) urbanized watersheds are devoid of fine sediment.

Examination of ecological changes along the urban – rural gradient have increased since McDonnell and Pickett's (1990) paper on an unexploited opportunity to study anthropogenic impacts. The present study on changes that occur within and adjacent to the stream channel contributes some key findings about the urbanization 'process'. The increasing percentage of urban land use within the watershed indicates relationships with not only water quality differences but also the stability of stream channels. While changes in stream habitat appear at 30% ULU, significant impacts occur once a watershed has greater than 45% ULU, at which point stream channels can not accommodate the power and intensity of impervious surface runoff. Homogenization of physical stream characteristics plays a vital role in the stability, resiliency, and overall integrity of the ecosystem, and may present too difficult a conservation challenge to overcome urbanization impacts.

Tables

Table 1. Stream sites and accessory information used to survey habitat complexity in the eastern Piedmont of Maryland. Site names were derived from the original MBSS site name, but reflect the year of sampling. Latitude and longitude are presented in decimal degrees. County abbreviations are BA = Baltimore, BC = Baltimore City, HA = Harford, HO = Howard, MO = Montgomery, and PG = Prince George's. The ULU (urban land use) and UCat (urban category) represent the percentage of urban land use in the upstream watershed. The river basins represented in this study were ANA = Anacostia, BUS = Bush, GUN = Gunpowder, PAT = Patapsco, PAX = Patuxent, and POT = Potomac. Watershed area upstream from each site is represented in hectares.

Site	Latitude	Longitude	County	ULU	UCat	Basin	Area
BYNU-105-2005	39.3388	-76.2017	HA	0.00	0-15	BUS	43
MPAX-107-2005	39.1166	-76.5772	HO	0.00	0-15	PAX	130
LOCH-112-2005	39.5250	-76.7907	BA	0.00	0-15	GUN	287
LOCH-114-2004	39.4948	-76.6847	BA	0.01	0-15	GUN	631
GWYN-102-2005	39.4062	-76.8241	BA	0.13	0-15	PAT	69
RKGR-119-2004	39.1685	-76.9720	HO	0.41	0-15	PAX	298
SBPA-108-2004	39.3479	-76.9166	HO	0.49	0-15	PAT	595
LPAX-115-2004	39.3047	-76.8978	HO	1.2	0-15	PAX	426
HO-125-2004	39.2640	-76.9550	HO	1.2	0-15	PAX	421

Site	Latitude	Longitude	County	ULU	UCat	Basin	Area
GWYN-112-2005	39.3955	-76.8114	BA	2.4	0-15	PAT	92
LOGU-106-2005	39.4499	-76.4533	BA	2.5	0-15	GUN	301
GWYN-105-2005	39.3888	-76.7709	BA	3.3	0-15	PAT	499
LIBE-101-2004	39.4697	-76.8593	BA	5.6	0-15	PAT	161
LPAX-112-2004	39.1519	-76.8866	HO	5.7	0-15	PAX	846
MO-137-2004	39.1190	-76.9120	MO	6.5	0-15	PAX	248
LIGU-102-2005	39.5067	-76.4293	HA	6.9	0-15	GUN	424
RKGR-107-2004	39.1384	-76.9702	MO	7.7	0-15	PAX	344
RKGR-106-2004	39.1804	-77.0701	MO	7.8	0-15	PAX	509
SENE-114-2005	39.2600	-77.2120	MO	13.1	0-15	POT	277
LWIN-112-2005	39.4438	-76.3331	HA	16.9	15-30	BUS	166
HO-114-2004	39.1560	-76.8190	HO	17.9	15-30	PAX	191
BYNU-109-2005	39.5489	-76.3513	HA	19.4	15-30	BUS	714
LIBE-102-2005	39.4532	-76.8326	BA	20.8	15-30	PAT	29
CABJ-109-2005	39.0220	-77.1920	MO	26.2	15-30	ANA	99
PATL-103-2004	39.1919	-76.7421	HO	27.3	15-30	PAT	908
ANAC-110-2005	39.0953	-76.9275	MO	27.8	15-30	ANA	171
LWIN-104-2005	39.4752	-76.3752	HA	29.6	15-30	BUS	78

Site	Latitude	Longitude	County	ULU	UCat	Basin	Area
LIGU-105-2005	39.4721	-76.3874	HA	31.7	30-45	GUN	74
BA-119-2005	39.2660	-76.7920	BA	34.4	30-45	PAT	211
LOCH-123-2005	39.4283	-76.5810	BA	35.6	30-45	GUN	218
HO-104-2005	39.1560	-76.8190	HO	38.1	30-45	PAX	191
JONE-109-2004	39.4067	-76.7280	BA	41.2	30-45	PAT	306
LPAX-116-2004	39.1872	-76.8614	HO	41.9	30-45	PAX	485
HO-120-2004	39.2740	-76.8410	HO	42.5	30-45	PAX	242
LIBE-107-2005	39.5739	-76.9867	CA	43.7	30-45	PAT	143
GWYN-107-2005	39.4572	-76.8018	BA	45.8	45-60	PAT	605
PATL-119-2004	39.2358	-76.7272	HO	47.5	45-60	PAT	399
GWYN-104-2005	39.3801	-76.8078	BA	47.8	45-60	PAT	188
LOCH-115-2005	39.4128	-76.5883	BA	48.6	45-60	GUN	51
PATL-105-2005	39.2470	-76.6660	BA	52.4	45-60	PAT	127
LWIN-120-2005	39.4382	-76.3162	HA	57.3	45-60	BUS	226
BA-117-2004	39.2620	-76.7110	BA	57.8	45-60	PAT	203
BIRD-101-2005	39.3800	-76.4880	BA	58.7	45-60	GUN	786
PATL-116-2005	39.2600	-76.7660	HO	61.4	>60	PAT	164
ANAC-116-2005	39.0226	-77.0307	MO	62.6	>60	ANA	906

Site	Latitude	Longitude	County	ULU	UCat	Basin	Area
LOGU-103-2004	39.4043	-76.5107	BA	64.2	>60	GUN	267
MO-127-2004	39.0960	-77.0130	MO	64.3	>60	POT	101
BACK-113-2004	39.3667	-76.5229	BA	64.86	>60	PAT	347
PAXU-105-2005	39.1042	-76.8884	PG	69.1	>60	PAX	95
CABJ-102-2005	39.0714	-77.1518	MO	73.0	>60	ANA	238
PATL-111-2004	39.2010	-76.7431	HO	73.6	>60	PAT	202
BA-128-2004	39.3420	-76.5140	BA	73.9	>60	PAT	387
BC-120-2004	39.3220	-76.6280	BC	74.9	>60	PAT	1161
LOGU-190-2005	39.2413	-76.3448	BA	74.9	>60	GUN	140
JONE-110-2004	39.3947	-76.6292	BA	75.4	>60	PAT	409
MO-126-2004	39.0710	-77.080	MO	80.8	>60	POT	202

Table 2. Analysis of variance (ANOVA) results for habitat variables measured in the 2004-2005 survey. A two-way ANOVA was performed using urban category (Ucat) and year to determine if significant differences (P < 0.05, bolded) exist across the land use gradient.

Parameter	Effect	Df	F-Value	P-Value
Temperature	Ucat	4, 46	1.18	0.33
	Year	1, 46	3.11	0.08
	Ucat*Year	4, 46	0.33	0.8535
Conductivity	**Ucat**	**4, 46**	**6.17**	**< 0.001**
	Year	1, 46	0.06	0.81
	Ucat*Year	4, 46	0.51	0.73
Dissolved Oxygen	Ucat	4, 46	1.71	0.16
	Year	1, 46	0.82	0.37
	Ucat*Year	4, 46	0.40	0.81
pH	Ucat	4, 46	0.83	0.51
	Year	1, 46	0.69	0.41
	Ucat*Year	4, 46	1.40	0.25
Extent of Riffle	Ucat	4, 46	1.55	0.20
	Year	**1, 46**	**5.05**	**< 0.05**
	Ucat*Year	**4, 46**	**2.77**	**< 0.05**
Extent of Run	Ucat	4, 46	1.21	0.32
	Year	1, 46	0.07	0.80
	Ucat*Year	4, 46	0.30	0.87
Extent of Pool	Ucat	4, 46	0.39	0.82
	Year	1, 46	3.64	0.06
	Ucat*Year	4, 46	1.18	0.33
Extent of Glide	Ucat	4, 46	0.46	0.76
	Year	**1, 46**	**3.95**	**0.05**
	Ucat*Year	4, 46	1.82	0.14
Erosion	**Ucat**	**4, 46**	**2.75**	**< 0.05**
	Year	**1, 46**	**6.84**	**< 0.05**
	Ucat*Year	4, 46	0.66	0.62
Maximum Height of Erosion	**Ucat**	**4, 46**	**6.01**	**< 0.001**
	Year	**1, 46**	**8.92**	**< 0.01**
	Ucat*Year	**4, 46**	**6.58**	**< 0.001**
Instream Rootwads	Ucat	4, 46	0.94	0.45
	Year	**1, 46**	**9.16**	**< 0.01**
	Ucat*Year	4, 46	1.13	0.35
Dewatered Rootwads	Ucat	4, 46	0.15	0.96
	Year	1, 46	2.33	0.13
	Ucat*Year	4, 46	0.13	0.97

Parameter	**Effect**	**Df**	**F-Value**	**P-Value**
Instream Woody Debris	Ucat	4, 46	1.22	0.31
	Year	**1, 46**	**9.12**	**< 0.01**
	Ucat*Year	4, 46	0.20	0.94
Dewatered Woody Debris	**Ucat**	**4, 46**	**3.29**	**< 0.05**
	Year	1, 46	0.70	0.41
	Ucat*Year	**4, 46**	**4.10**	**< 0.001**
Undercut Banks	Ucat	4, 46	1.82	0.14
	Year	1, 46	0.51	0.48
	Ucat*Year	4, 46	1.12	0.36
Bank Stabilization	**Ucat**	**4, 46**	**6.01**	**< 0.001**
	Year	1, 46	0.07	0.80
	Ucat*Year	4, 46	0.21	0.93
Debris Jams	Ucat	4, 46	1.06	0.39
	Year	1, 46	2.50	0.12
	Ucat*Year	4, 46	1.47	0.23
Bar Formation	**Ucat**	**4, 46**	**3.23**	**< 0.05**
	Year	1, 46	0.00	0.97
	Ucat*Year	4, 46	1.41	0.25
Tributary	Ucat	4, 46	0.53	0.71
	Year	1, 46	0.25	0.62
	Ucat*Year	4, 46	0.39	0.81
Pipe	**Ucat**	**4, 46**	**2.56**	**< 0.05**
	Year	1, 46	2.17	0.15
	Ucat*Year	4, 46	1.27	0.29
Gabion	Ucat	4, 46	2.29	0.07
	Year	1, 46	1.63	0.21
	Ucat*Year	4, 46	2.29	0.07
Gabion and Bank Stabilization	**Ucat**	**4, 46**	**5.78**	**<0.001**
	Year	1, 46	0.58	0.45
	Ucat*Year	4, 46	0.67	0.62
Bridges	Ucat	4, 46	0.37	0.83
	Year	1, 46	0.02	0.89
	Ucat*Year	4, 46	0.13	0.97
Discharge	**Ucat**	**4, 46**	**2.62**	**< 0.05**
	Year	1, 46	0.92	0.34
	Ucat*Year	4, 46	0.62	0.65
Average Depth	Ucat	4, 46	0.64	0.64
	Year	1, 46	0.02	0.88
	Ucat*Year	4, 46	1.16	0.34
Average Width	Ucat	4, 46	0.33	0.86
	Year	1, 46	0.01	0.93
	Ucat*Year	4, 46	0.81	0.53
Width:Depth Ratio	Ucat	4, 46	0.55	0.70
	Year	1, 46	0.61	0.44
	Ucat*Year	4, 46	0.74	0.57

Parameter	Effect	Df	F-Value	P-Value
Average Shading	Ucat	4, 46	0.09	0.99
	Year	1, 46	0.89	0.35
	Ucat*Year	4, 46	2.04	0.10

Table 3. Linear extent (m) of channel subunits across the urban – rural gradient. Each value represents the mean ± SEM. *Ucat = urban category*.

Ucat	N	Riffle	Run	Pool	Glide
0-15 %	19	19.9 ± 2.43	27.1 ±3.22	23.3 ± 3.20	7.9 ± 2.33
15-30 %	8	22.9 ± 4.71	17.5 ± 5.46	30.8 ± 7.68	5.6 ± 4.68
30-45 %	8	22.3 ± 3.60	26.3 ± 2.53	25.1 ± 3.11	3.8 ± 2.18
45-60 %	8	14.8 ± 6.69	21.1 ± 6.53	29.9 ± 4.67	15.4 ± 8.33
> 60 %	13	13.2 ± 2.73	31.9 ± 3.69	20.9 ± 3.86	11.2 ± 3.60

Table 4. Stepwise multiple regression equations explaining variance in habitat parameters. All habitat variables were utilized in this analysis, allowing forward and backward selection into the final equation at $P = 0.05$. Variables are listed in order of their contribution to the final equation, with the greatest contribution first.

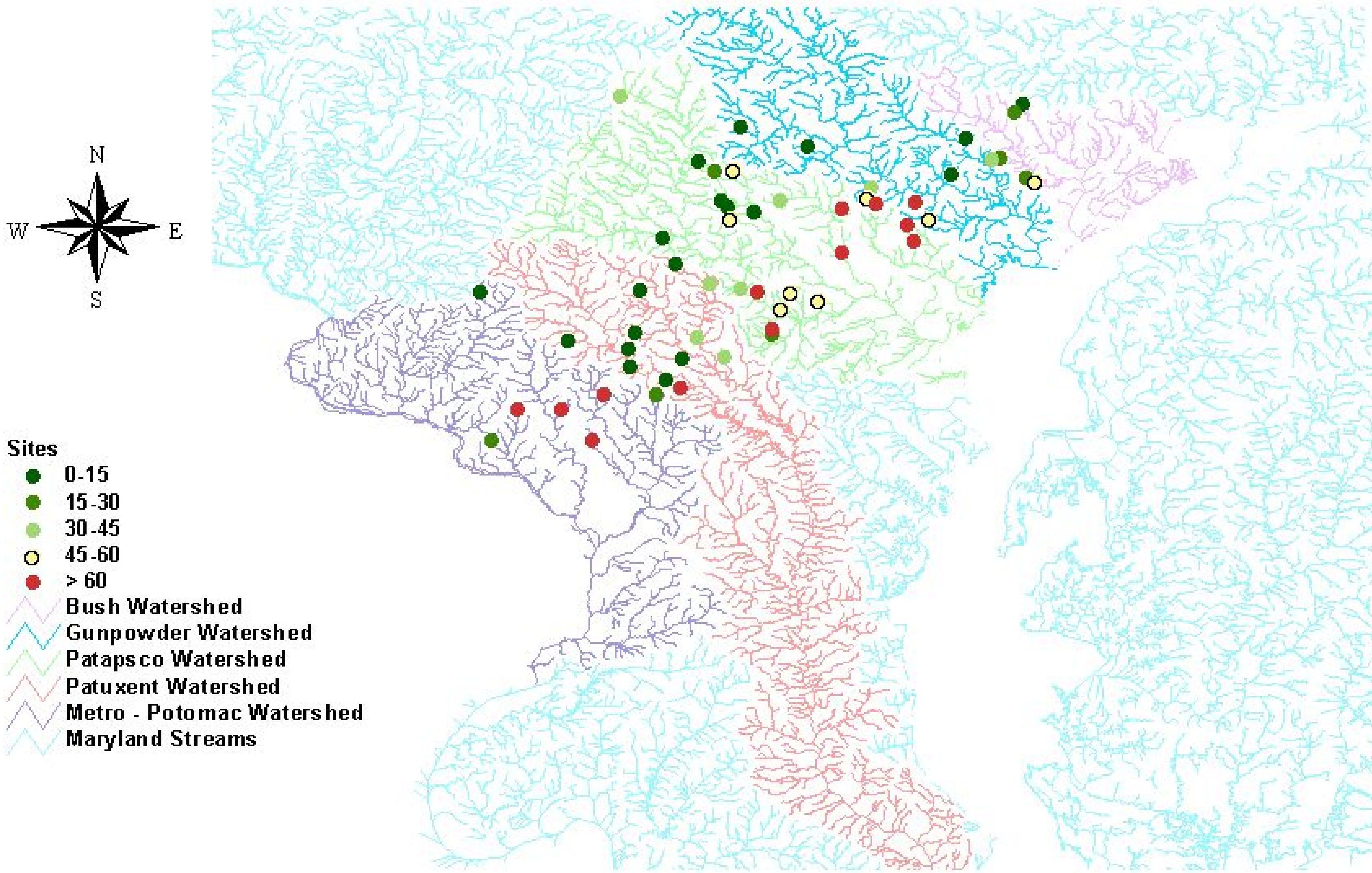
Parameters	**Adj-R^2**
Erosion = 75.82 + 1.48(Impervious Surface) – 0.87(Riffles) + 2.48(Dewatered Rootwads) – 1.27(Gabion)	0.44
Conductivity = -1.17 + 0.01(Impervious Surface) + 0.18(pH) + 0.004(Pools)	0.44

Table 5. Bar substrate composition. Bars were characterized by the presence of silt, sand, gravel, cobble, boulders or bedrock. Values in the heading represent the total number of sites with that substrate, while the values below represent the frequency of presence in streams in each urban category. *Ucat = urban category*.

Ucat	**Silt (18)**	**Sand (55)**	**Gravel (54)**	**Cobble (48)**	**Boulders (11)**	**Bedrock (3)**
0-15%	0.42	1.0	1.0	1.0	0.05	0.00
15-30%	0.25	1.0	1.0	0.63	0.00	0.00
30-45%	0.13	1.0	1.0	1.0	0.38	0.13
45-60%	0.50	0.88	0.75	0.38	0.25	0.13
>60%	0.23	1.0	1.0	1.0	0.39	0.08

Figures

Figure 1. Map of stream sites surveyed for habitat complexity study. The legend indicates the site membership to urban categories (0-15, 15-30, 30-45, 45-60, and > 60% ULU), and in which watershed each site was found.



N
W
E
S
Sites
0-15
15-30
30-45
45-60
> 60
Bush Watershed
Gunpowder Watershed
Patapsco Watershed
Patuxent Watershed
Metro - Potomac Watershed
Maryland Streams

Figure 2. Stream conductivity (mS/cm) across the urban- rural gradient, expressed in urban categories (0-15, 15-30, 30-45, 45-60 and > 60% urban land use). Each bar represents the mean of streams sampled in 2004 and 2005 plus the standard error of the mean. Homogeneous groups are indicated the same letters.

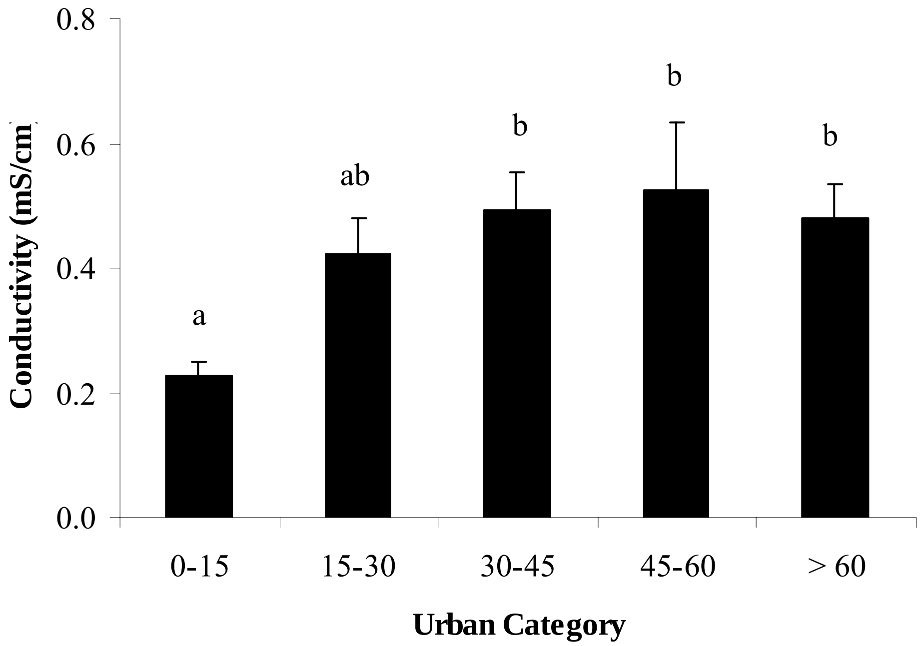


Figure 3. Extent of bank stabilization by boulders, cobble, fiber netting or other man-made structures across the urban-rural gradient. Each bar represents the mean of streams sampled in 2004 and 2005 plus the standard error of the mean. Streams within the 0-15% ULU category did not exhibit any anthropogenic bank stabilization practices. Homogeneous groups are indicated by the same letters.

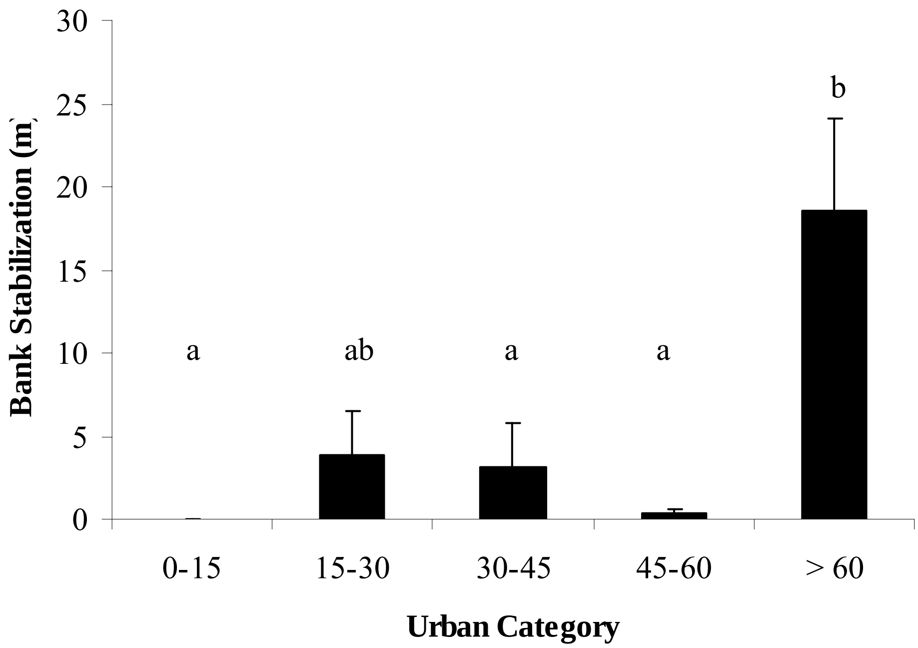


Figure 4. Extent of engineered structures found on streambanks along the urban – rural gradient. Each column represents the mean of streams sampled in 2004 and 2005 plus the standard error of the mean. Streams within the 0-15% ULU category did not exhibit any anthropogenic bank stabilization practices. Homogeneous groups are indicated by the same letters.

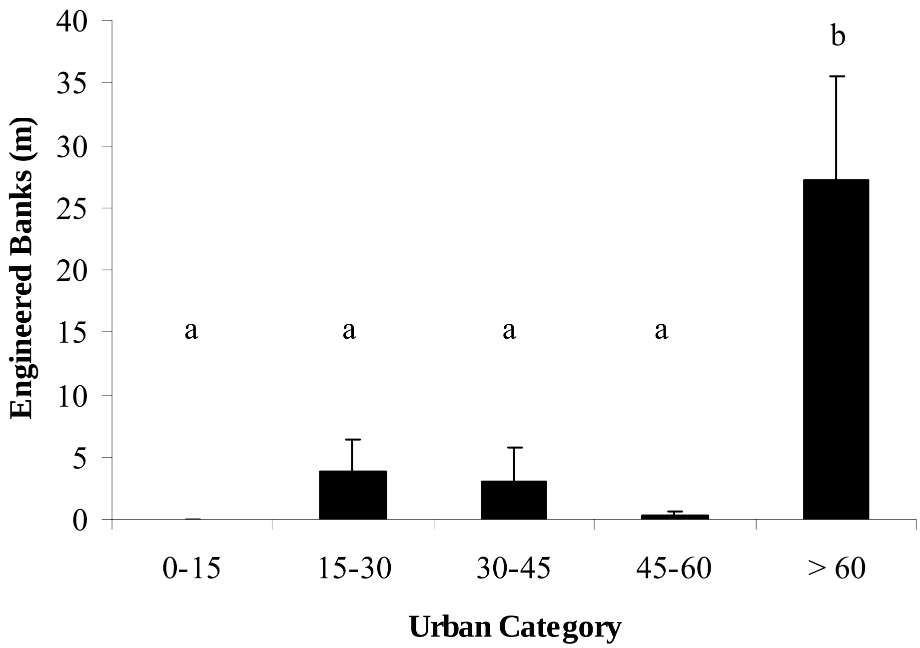


Figure 5. Linear extent of bars (m) formed in the stream channel across the urban – rural gradient. Each column represents the average total length of all bars found in the channel, including those on left and right bank as well as those found mid-channel, plus the standard error of the mean. Homogeneous groups are indicated by the same letters.

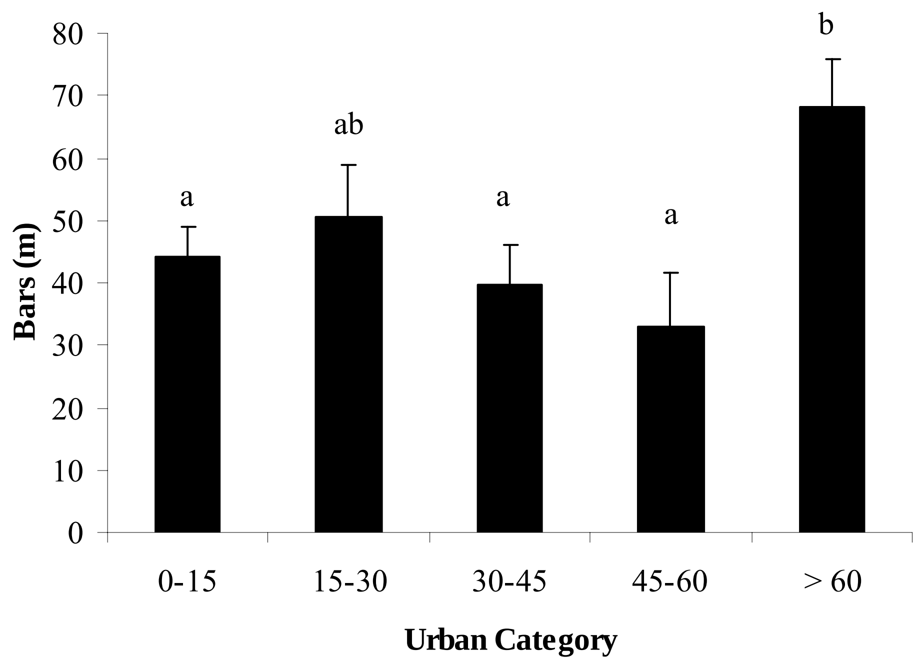


Figure 6. Total number of dewatered woody debris along streambanks across the urban – rural gradient. Bars representing the mean of each category plus the standard error of the mean are split into the year surveyed. Homogeneous groups are indicated by the same letters for 2004 data only.

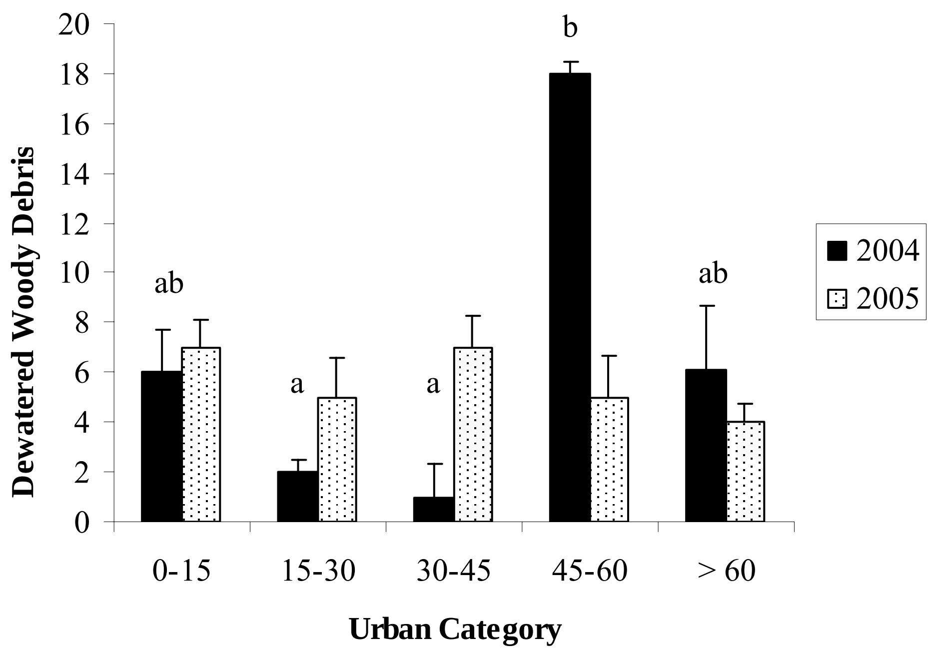


Figure 7. Maximum height of erosion (m) along streams across the urban – rural gradient. Bars represent the mean height of erosion plus the standard error of the mean for each year. Homogeneous groups are indicated by the same letters in 2004 data only.

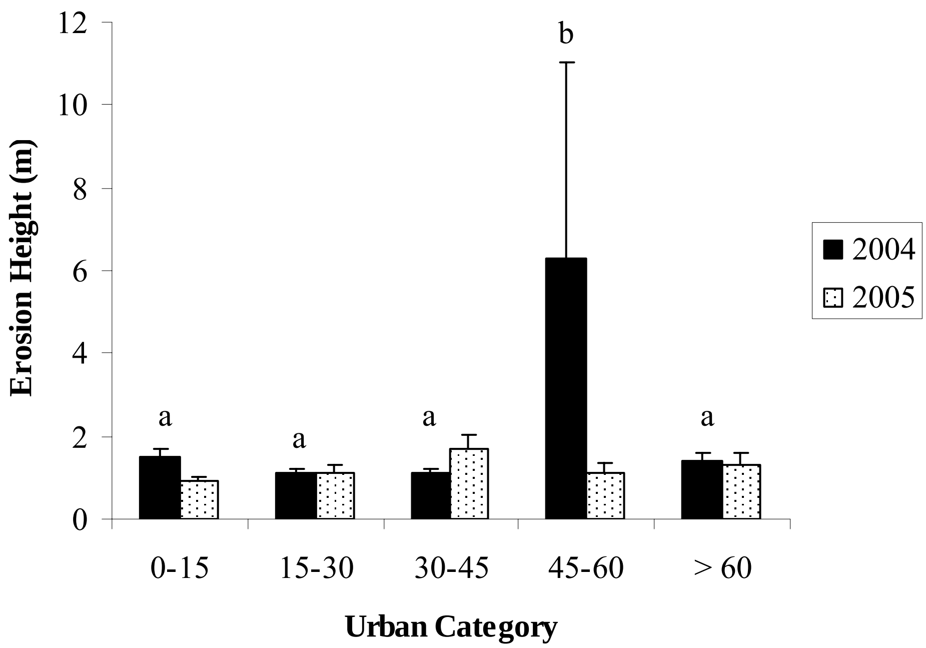


Figure 8. Linear relationship between % impervious surface and % urban land use (ULU) within a watershed. Least squares regression suggests that ULU predicts 85% of the variation in % impervious surface ($P < 0.0001$; n = 56).

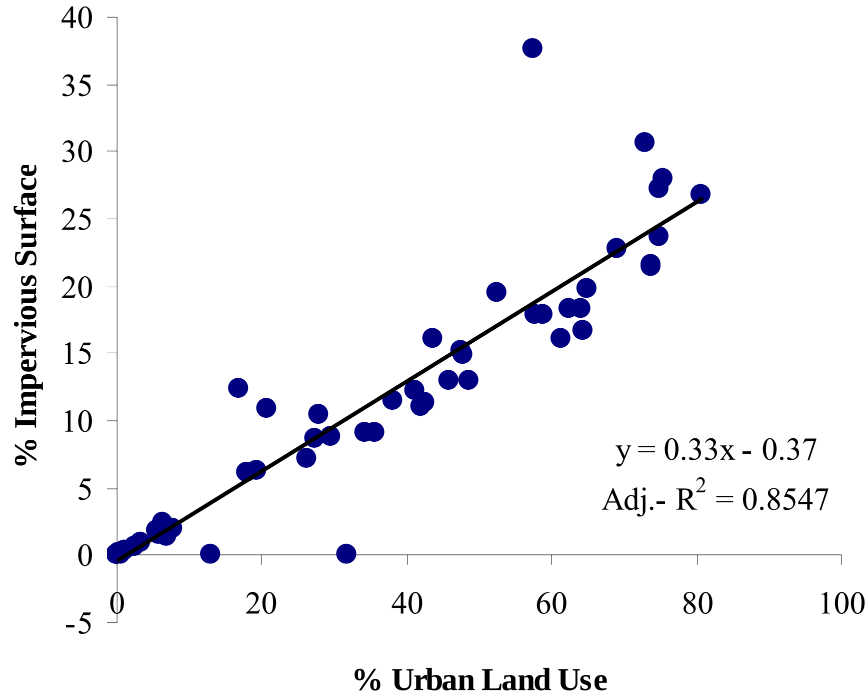


Figure 9. Linear relationship between impervious surface and the linear extent of eroded banks (m). Percent impervious surface within the watershed predicts 12% of the variance in eroded banks across the urban – rural gradient ($P < 0.001$; n = 56).

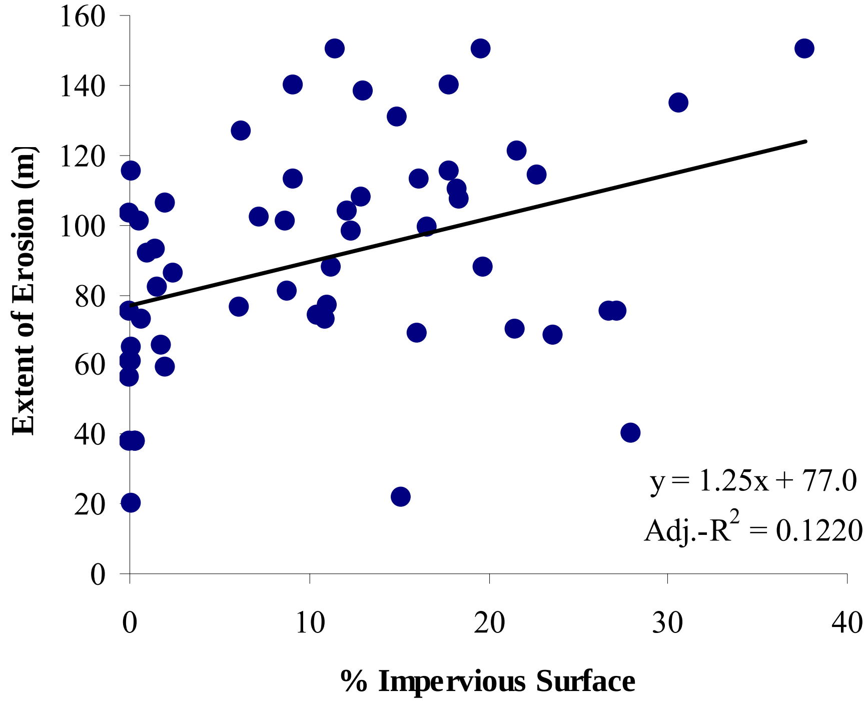
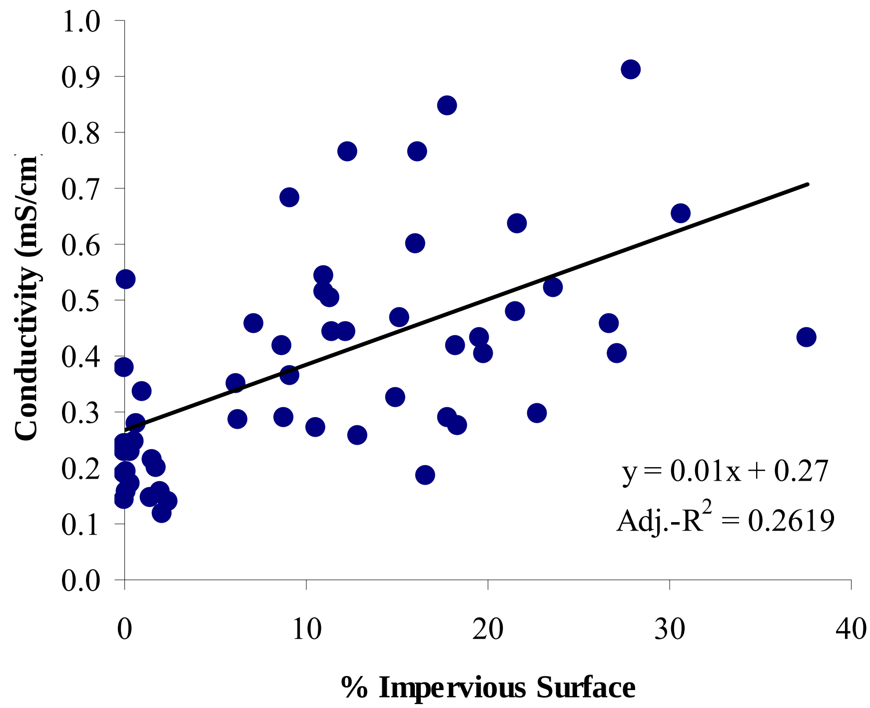


Figure 10. Linear relationship between % impervious surface and conductivity (mS/cm) of the stream water. Percent impervious surface within the watershed predicts 26% of the variance in conductivity across the urban – rural gradient ($P < 0.0001$; n = 56).



Chapter 4: Habitat selection by stream cyprinids across the urban – rural gradient: implications for stream restoration

Abstract

The Mid-Atlantic region is a hot spot for stream habitat restoration in degraded watersheds yet few studies have determined whether the fish assemblage would respond to restoration practices. I tested effects of instream habitat enhancement through fish selection response using three treatments (woody debris - LWD, shade - SH, and both - SHWD) in first order urban (> 60% urban land use, ULU), suburban (27-46% ULU), and rural (< 15% ULU) eastern Piedmont streams in Maryland (n = 36). Twenty meter block-netted experimental segments were split into combinations of one enhancement section (10 m) paired with a control section (10 m). Fish were removed by double-pass electrofishing, treatments were constructed, and only *Rhinichthys atratulus* and *Semotilus atromaculatus* were replaced into the center of the segment. For 6 h the fish were allowed to range freely between sections, then treatment and control sections were separated by a blocknet and fish were retrieved and tallied. Habitat selection was significantly different between rural SHWD vs. LWD, and between SHWD and SH in suburban fish ($P < 0.05$). Fish total length differed significantly between urban, suburban, and rural fish, where urban fish were the smallest ($P < 0.05$). CKB who selected the treatment were significantly larger than in the control section ($P < 0.05$). Size-dependent habitat segregation may occur as a result of intraspecific competition. Rural and suburban fish recognized and selected the most complex stream habitat

enhancements, yet urban fish most commonly selected SH. Thus, increasing the amount of overhead cover in urban stream channels would be beneficial for fish populations when implementing stream restoration practices.

Introduction

Impairment of running waters due to a variety of anthropogenic influences both on land and water is recognized as an international issue (Schlosser 1991, Paul and Meyer 2001, Gergel et al. 2002, Groffman et al. 2003, O'Neill et al. 1997, Poff et al. 1997, Richards et al. 1996, Walsh et al. 2005). One-third of US rivers are considered to be polluted or impaired in some way (USEPA 2000). Maryland has the highest density of stream and river restoration projects in the country (Bernhardt et al. 2005a). Instream habitat improvement (~ 40%), water quality (~ 30%), and bank stabilization (~ 4%) are the top three types of stream restoration efforts in Maryland, costing approximately $5.6 billion per 1000 km (Bernhardt et al. 2005b). Other restoration practices include aesthetics, channel reconfiguration, dam removal, fish passage, floodplain reconnection, flow modification, instream species management, land acquisition, and riparian and stormwater management (Bernhardt et al. 2005b, Hassett et al. 2005). There has been a marked increase in the number of restoration projects across the nation since 1990, and within the Chesapeake Bay watershed since 1995 (Bernhardt et al. 2005b, Hassett et al. 2005). Unfortunately only a small percentage of projects include some type of pre- or post-restoration monitoring, many times due to a lack of funds. Among those projects that were monitored, installation of fish ladders to provide fish passage, and floodplain reconnection practices were most common, with stormwater management monitoring close behind (Hassett et al. 2005).

The goals of most restoration projects vary spatially and temporally, based on major stressors within the watershed, species at risk, and the level to which pre-disturbance conditions are expected (Booth 2005). Physical habitat changes or channel morphology objectives may be set, with hopes that stream biota will return (The Field of Dreams hypothesis – Palmer et al. 1997). Interestingly, many stream restoration efforts lack clearly defined biotic objectives, and without proper monitoring, it is difficult for managers to assess effective and successful projects (Booth 2005, Palmer et al. 2005). Post-restoration monitoring of the biotic community (usually fish, macroinvertebrates, and plants) must be appropriately planned to determine successful short- and long-term design enhancement as well as successful endpoints (Booth 2005). For example, sampling for macroinvertebrates and fish soon after the project completion may present great variability in species richness and abundance depending on the type of impairment, restoration practice, and length of disturbance during project construction (Shields et al. 2003). Conversely, biotic integrity monitoring years after restoration may deem the project a failure due to a lack of species improvement (Bond and Lake 2005, Eklöv et al. 1998, Moerke et al. 2004a). Thus, monitoring should occur on a more frequent basis (pre and post construction) to fully understand its implication.

Booth (2005) emphasizes that both short and long-term enhancement of streams may be reached if the actions address the appropriate elements of restoration. This temporal difference in reaching successful endpoints is important to distinguish whether or not the project goals are feasible to begin with. Short-term enhancements serve acute problems that can be addressed with relatively immediate solutions, while long-term enhancements become self-sustaining to the stream ecosystem (Booth 2005). Depending on the type of restoration practice (e.g. fish passage vs. instream habitat improvement), the re-establishment of stream biotic integrity should be expected on different time scales. Thus, planning of post-restoration monitoring should be evaluated with temporal goals in mind.

Urbanization effects on small stream ecosystems have been increasingly studied, providing new insights on biological composition, and physiochemical and ecosystem processes (Morgan and Cushman 2005, Paul and Meyer 2001, Meyer et al. 2005, Riley et al. 2005, Roy et al. 2003, Walsh et al. 2005). The urban stream syndrome, named by Meyer et al. (2005), is defined by a set of characteristics that describe the ecological degradation of the above ecosystem patterns and processes (Walsh et al. 2005). Streams exhibiting the urban stream syndrome are commonly found in watersheds with high percentage of urban land use and impervious surfaces. Comparative studies of land use and ecological patterns have followed a gradient conceptual framework of rural to urban environmental settings (McDonnell and Pickett 1990) and have become common in both experimental and restrospective research (this study, Limburg and Schmidt 1990, Morgan and Cushman 2005, Fraker et al. 2002, Wear et al. 1998). Meanwhile, stream ecosystem restoration research has commonly been performed in urban watersheds, paired with forested, rural watersheds. Therefore, a study of potential restoration outcomes across multiple land use categories would provide a better outlook of community and ecosystem changes.

Instream habitat enhancements include a variety of techniques, but addition of large woody debris (LWD) to deflect flow and create refugia for macroinvertebrates and fish is most prevalent. Lemly and Hilderbrand (2000) experimentally added LWD to a small Apppalachian stream to test whether relationships exist between benthic detritus, macroinvertebrates and LWD. In Australia, LWD was incorporated into a sand-bottomed stream to increase channel complexity and fish refugia, particularly during low flows (Bond and Lake 2005). Roni and Quinn (2001) examined fish movement patterns between restored (complex channel - LWD placement) and unrestored (simple channel) stream reaches to determine if fish would move towards higher quality habitat.

Correlative studies between habitat structure, complexity and associated fish assemblages have governed the design of many of these experimental stream projects (Gorman and Karr 1978, Inoue and Nunokawa 2002, Thévenet and Statzner 1999, Matthews et al. 1994). Yet, few restored reaches have (1) indicated a successful spatially-implicit biotic response; and (2) been able to quantify improved assemblages as a result of habitat enhancement. Only two studies were able to suggest that fish actively preferred and selected habitat enhanced by LWD placement over the unrestored reach (Giannico 2000, Roni and Quinn 2001). Giannico's (2000) experimental manipulations also involved dispersal of food along with increased LWD though, and indicated that food was the dominant attraction to the habitat patch. Both studies were performed in the Pacific Northwest on juvenile coho salmon, cutthroat trout and/or steelhead. No studies have been conducted on non-salmonid fish species. Moerke et al. (2004a) found that fish biomass but not abundance increased in restored meanders above unrestored reaches; however, the authors were neither specific about spatial patterns nor species captured.

Urbanization age greatly influences the stream community composition. Few urbanized watersheds on the east coast where restoration projects have been implemented have salmonid species present. Most of the fish species found in abundance in

Maryland's urban streams are generally pool-dwelling but are not as habitat specific as many salmonid species (Jenkins and Burkehead 1993). Thus, although habitat preference may be well-known, it is inappropriate to assume that these pollution-tolerant, omnivores would actively seek out and select 'enhanced' stream reaches that have been mechanically restored.

Evidence has revealed the presence of a knowledge gap between the expected biotic response and the actual response to stream restoration efforts. To examine habitat preference and selection responses, I questioned whether fish would select enhanced habitat patches mimicking local-scale stream restoration efforts over a short amount of time. To answer this question, I tested the following hypotheses. Given the choice of enhanced versus unenhanced habitat within each stream site, I hypothesized that fish would select the enhanced habitat greater than 50% of the time in all stream/land use categories. Specifically, I hypothesized that 1) fish in rural streams would select shade or combined shade and large woody debris more than just woody debris, 2) fish in suburban streams would respond better to a combination of large woody debris and shade than other types of enhancement, and that 3) urban fish would not select any one enhancement more than another.

I also questioned whether fish size played a role in habitat use and selection in this experiment. First, I hypothesized that the lengths of fish found in the control and treatment sections would differ. Secondly, I hypothesized that fish total length (TL) would differ among urban, suburban, and rural streams, with urban fish being the smallest.

Methods

My study was conducted across three urban land use (ULU) categories, rural (<15% ULU), suburban (27-46% ULU), and urban (>60% ULU) throughout the Baltimore-Washington corridor (12 sites per category; Table 1, Figure 1). Within each ULU category, I tested the effects of three different stream habitat treatments within 20 m channel segments, replicated four times (n = 36; Appendix II). Blacknose dace (BND) *Rhinichthys atratulus* and creek chub (CKB) *Semotilus atromaculatus* were selected for use in this study due to their presence in stream networks of rural, suburban, and urban watersheds. BND and CKB are considered pollution-tolerant fish species, and their ubiquity makes them excellent organisms for this type of comparative study.

Study sites

First order stream sites in the eastern Piedmont physiographic province were selected for this study from the MUF database (see Chapter 3) created from the 1995-1997 and 2000-2004 Maryland Biological Stream Survey dataset. Site criteria included the percent urban land use found in the upstream watershed, the presence of BND and CKB, and channel width. In order for the treatments to have a potential effect on habitat selection, streams less than 4 m wide were studied. The 36 stream sites involved in this study were located in Harford, Baltimore, Carroll, Howard, and Montgomery counties in Maryland during June, July and August of 2004-2005 (June = 10, July = 11, August = 15; 2004, n = 9; 2005, n = 27; Table 1). These sites were found in the Bush (n = 2), Gunpowder (n = 8), Patapsco (n = 14), Patuxent (n = 6), and Potomac (n = 6) River basins (Table 1, Figure 1). One site (HO-120-2004) was repeated in 2005 in a different segment of the stream reach. This site was the first experiment done in 2004, and due to silty stream bottom conditions, recapture was only 29% efficient and more species were collected at the close of the experiment than at the start. Therefore, data from 2004 was replaced with the 2005 experiment. The position of each site was taken using a GPS and recorded.

Habitat patch experiment

An experimental segment within each stream was selected based on a brief survey of channel characteristics and frequency of channel subunits. I selected segments that were 20 m in length, characterized primarily by pool habitat. Some sites had riffle or run habitat present within the 20 m; however, in these cases, the experimental segment was selected to include pool habitats at each end. Water quality measurements were made once at each site above the 20 m segment. A Hydrolab® Quanta® was placed in the middle of the stream channel to collect stream water temperature (°C), pH, dissolved oxygen (mg/L), and specific conductivity (mS/cm) measurements. Depth (m) and velocity (m/s) were measured at regular intervals across the width of the stream (m) to estimate discharge (m^3/s).

Once an experimental segment was selected, blocknets were placed at each end and secured with cobble along the stream bottom and with stakes along the streambank. Fish collection was performed using double-pass electrofishing (Smith-Root® model 12 backpack battery electrofisher) in an upstream direction. Electrofisher voltage was adjusted to the lowest possible setting, based on the measured conductivity of the stream water, in order to reduce potential injury from repeated exposure. Immobilized fish were collected and placed into 19-liter buckets filled with stream water. Fish from each pass were identified and tallied by species. All BND and CKB were held in a bucket with aerated water during the second pass while all non-target fish species were released downstream of the segment. The stream was allowed to settle and then a second pass was performed. Fish were collected in a new bucket and subsequently identified and tallied. BND and CKB individuals from both passes were combined and held in aerated water while the treatment was constructed in the stream. Approximately 40-50 BND and CKB (combined) and a maximum of 60 individuals were used in the experiment. If the total number of fish collected in 20 m was less than 40, up and/or downstream reaches were electrofished until the appropriate number of fish had been collected.

The experiment consisted of three stream enhancement treatment combinations. The 20 m experimental segment was divided into two 10 m sections to which one of three treatments were applied (Figure 2). One treatment involved the addition of three large woody debris (LWD) pieces, which were used to represent structure in the stream channel. In a second treatment, the stream was enhanced by providing shade (SH) through overhead cover in which two large tarps were secured over the stream channel. The third treatment was a combination of the LWD and SH (SHWD), and the fourth was a control in which no stream habitat enhancement was added. The use of LWD, SH, or SHWD was randomly chosen prior to the stream visit and paired with the control. The position of the treatments was also randomly chosen (upstream or downstream) within the experimental segment to eliminate any blocknet effects during the experiment.

Similar sized LWD was placed mostly submerged, in a downstream alternating weir formation such that logs were angled laterally into the water in the direction of flow (Appendix III). Each tarp was 4 m x 5 m in size and secured to stream banks using large cobble, rebar, or tied to trees with rope. The tarps were positioned such that they hung within 1 m of the stream water surface, providing a protective reduction in ambient light to the water column. When the SHWD treatment was implemented, the woody debris was positioned in the stream channel first, and then the tarps were suspended and secured overhead. Once assembly of the treatment was completed, all captured BND and CKB were replaced in the middle of the 20 m segment from which they were drawn, essentially along the treatment boundary.

From this point, the fish were given 6 h to readjust, relocate, and select the stream habitat area of their choice. At the end of the habitat selection time period, a third blocknet was discretely and quickly placed across the stream channel at the 10 m position to keep the fish separated within their selected habitat (Figure 2). Once the blocknet was secured, the treatments were removed from the stream channel and the two experimental sections were sampled via double-pass electrofishing. Fish were collected in separate buckets, identified, measured for total length (TL), and counted for each section after first pass. Once the stream water settled, the same method was applied for the second pass fish capture. Finally, the blocknets were removed and fish were replaced in the stream. The number of fish collected in each section at the end of the experiment was used for comparison and evaluation of habitat selection across treatment type and landuse category.

The experimental segment was characterized by measuring stream width at the 0, 10, and 20 m positions, and a stream map of physical habitat was drawn. Position of all LWD (including that from LWD and SHWD if present), rootwads, channel subunit presence (pool, glide, run, riffle), bar formation, dominant substrate type, debris jams, streambank characteristics, and any additional miscellaneous notes were recorded for each site.

Experiments were conducted at about the same time each day (first pass – 9:00am, second pass – 9:30am, setup – 10:00am, finished – 5:00pm). The experiment was run on mostly fair weather days, although sampling at three sites were complicated by impending afternoon thunderstorms. In two cases (MO-127-2004 and PATL-103-2005), the experiment was ended an hour early in order to avoid heavy downpours. In a third case (CABJ-102-2005), a 20 min light shower during the fourth hour of the experiment caused stage height to rise and strained the blocknets. However, in each of these cases, recapture efficiency was high (MO – 94%, PATL – 95%, CABJ – 103%). Finally, equipment failure occurred at one experimental site, thus only allowing a single-pass of electroshocking (BA-126-2005; recap efficiency – 79%).

Recapture efficiency was very high for the majority of experiments. The average recapture for sites treated with LWD was 104 ± 7%. At sites treated with only SH, I recovered 108 ± 5%, while at sites treated with SHWD, I recovered 98 ± 5% at the close of the experiment. The minimum and maximum recapture efficiencies were 67% (rural site) and 160% (urban site).

Calculations and statistical analyses

Species richness and relative abundance were estimated using fish collected in two passes at the outset of the experiment. Species richness was calculated by tallying the number of species found, while relative abundance was estimated by summing the number of fish individuals of all species collected within the experimental segment. Sampling/recapture efficiency was calculated by dividing the post-experiment capture (total number of fish recaptured) by the pre-experiment capture (number of fish put into the experimental reach).

Comparisons of treatment and ULU category were made to determine if fish selected the experimentally enhanced stream section over the control (not enhanced). The experimental design required that standardization of the abundance data for each site, because the number of fish used in each experiment varied. I used the response variable treatment proportion, which equaled the number of fish (BND and CKB combined) collected in the treatment divided by the total number of fish collected at the close of the experiment. Since fish had the ability to freely roam between the enhanced and control sections, the null hypothesis was that 0.5 of the fish would be found in the treatment section and 0.5 would be found in the control section. The alternative hypothesis stated that different percentages of fish were found in the treatment and control sections. A two-way ANOVA was performed on the treatment proportion across treatments and ULU categories to determine if treatment and land effects existed. I also tested for downstream and upstream treatment bias on the response data. Individual species responses were run through the same experimental effects analysis as the combined data to determine if one species was responsible for specific habitat or treatment selection. Treatment proportion was assessed across all treatments and ULU categories to determine if treatment or land effects were present in the data.

Fish total lengths were analyzed using a randomized complete block split-plot design, blocking by land use category for each species to test the first hypothesis. The whole-plot factor was the type of treatment applied (LWD, SH or SHWD) and the split plot was the section the fish was found in (control, treatment). Fish length data were also analyzed with a one-way ANOVA for each species to detect differences among land use categories.

Supplementary data on environmental stream conditions were analyzed for differences across the ULU categories to determine if species richness and abundance as well as treatment effects varied in response to stream integrity. All statistical analyses were conducted using SAS (SAS Institute 1999). Data were checked for conformation to a normal distribution. Type I error was controlled when multiple comparisons were made using Tukey's adjusted *P*-values. Statistical differences among the data were reported at $\alpha = 0.05$ level.

Results

Across the land use gradient, rural streams had the greatest species richness, followed by suburban and urban streams (Figure 3), and there was a significant land use effect on fish species richness ($F = 6.6$; $df = 2, 33$; $P < 0.01$). Rural richness was significantly higher than urban stream fish richness ($t = 3.61$; $P < 0.05$). Suburban richness was not different from urban ($t = 2.15$; $P = 0.10$) and or rural richness ($t = 1.46$; $P = 0.32$). Abundance of fish found in the 20 m segment was also analyzed. In this case, there was no difference in fish abundance across the three land use categories ($F = 0.2$; $df = 2, 33$; $P = 0.84$; Figure 4). Finally, I used the species richness and relative abundance data to test the effects of conducting this experiment in two different years. Neither richness nor abundance differed (richness $F = 1.0$; $df = 1, 30$, $P = 0.33$; abundance $F = 3.1$; $df = 1, 30$; $P = 0.09$) between years, although abundance was a little higher in 2004 streams (92.3 ± 16.7 vs. 2005: 58.9 ± 8.77).

Among all the water quality and discharge measurements taken, temperature was the only parameter that suggested a land use effect (F = 5.6; df = 2, 33; P <0.01). Temperature was significantly higher in urban streams than suburban (t = -2.69; P < 0.05) and rural streams (t = -3.07; P < 0.01; Table 3). Conductivity values were also highest in urban streams, however there was no significant land use effect (F = 2.69; df = 2, 33; P = 0.08; Table 3). A gradient in dissolved oxygen and discharge values appeared, with the highest values in rural streams and lowest in urban streams; however, there was no statistically significant difference among the stream categories (F = 2.98; df = 2, 33; P = 0.06 and F = 2.40; df = 2, 33; P = 0.10, respectively). There was no discernible pattern in pH across ULU categories (F = 1.31; df = 2, 33; P = 0.28; Table 3).

Stream channel morphology parameters were also compared across ULU categories. Maximum depth in the treatment and control sections of each experiment as well as the average width of the channel for each stream were assessed. The absolute value of the difference between the control and treatment section depth suggested no significant land use effect (F = 1.34; df = 2, 20; P = 0.29). Although urban channels were slightly wider than suburban and rural channels, there was no indication that stream channels were significantly wider or narrower in any one ULU category (F = 2.18; df = 2, 20; P = 0.13; Table 4).

When given the choice of enhanced and unenhanced habitat, fish responded positively to treatments relative to controls (F = 4.95; df = 2, 27; P < 0.05). There were no significant effects of land use (F = 1.06; df = 2, 27; P = 0.36), nor were there significant interaction effects (F = 1.33; df = 4, 27; P = 0.28). Difference of least squares means indicated that the overall response to LWD was significantly different than the response to SHWD ($t = 3.15$; $P < 0.01$).

Within land use categories, fish responded better to some treatments than others. In rural streams, the response to SHWD was significantly greater than to just LWD ($t = 2.64$; $P < 0.05$). However, suburban fish responded significantly higher to SHWD than to SH ($t = 2.09$; $P < 0.05$). In rural and suburban streams, the response to the SHWD was greatest among all three treatments, while the greatest response in urban streams was to the SH treatment (Figure 5). The lowest response in rural and urban streams was to LWD, while the lowest response in suburban streams was to SH alone (Figure 5).

The data were subjected to ANOVA using treatment and position of treatment to test for upstream or downstream bias in fish response. There was still a treatment effect ($F = 5.04$; df = 2, 30; $P < 0.05$); however, there was no effect of treatment position ($F = 0.08$; df = 1, 30; $P = 0.77$), nor was there an interaction effect ($F = 1.09$; df = 2, 30; $P = 0.35$). To isolate species responses, the treatment effect analysis was run again using only BND or CKB in the form of percent treatment. Neither BND nor CKB indicated a significant response to the treatment or land use main effects, or to the land use–treatment interaction effect (Table 2).

Differences in total length (TL) of fish recovered in enhanced and unenhanced habitat sections varied by species. TL did not differ between BND found in the control and treatment sections of this experiment ($F = 1.2$; df = 1, 759; $P = 0.27$), nor were there any significant effects of treatment ($F = 0.4$; df = 2, 4.11; $P = 0.72$) or treatment – section interactions ($F = 2.8$; df = 2, 760; $P = 0.06$). However, there were significant section effects ($F = 7.6$; df = 1, 394; $P < 0.01$) and treatment–section interaction effects for CKB

($F = 3.2$; df = 2, 393; $P < 0.05$). CKB found in the treatment section of the experiment were larger than individuals in the control section (78 ± 5.5 mm vs. 71 ± 5.5 mm, respectively). There was no effect of treatment on CKB total length ($F = 1.1$; df = 2, 3.7; $P = 0.43$).

When fish TL was compared across ULU, significant differences were found in both BND ($F = 13.2$; df = 2, 766; $P << 0.001$) and CKB ($F = 9.4$; df = 2, 399; $P << 0.001$). Urban BND were significantly smaller than both suburban ($t = 4.03$; $P <<0.001$) and rural fish ($t = 4.47$; $P <<0.001$; Figure 6); but suburban and rural BND were not significantly different in length ($t = 0.61$; $P = 1.00$). Urban CKB were also smaller than both suburban ($t = 3.51$; $P < 0.01$) and rural individuals ($t = 4.25$; $P << 0.001$; Figure 5). Comparison of rural and suburban CKB did not indicate a significant difference in total length ($t = 1.25$; $P = 0.63$).

Discussion

Stream restoration projects with goals of increased biotic diversity, habitat use, and channel complexity are rarely designed with resident fish populations in mind. Since altered flow regimes are considered the acute stressors in urban systems (Paul and Meyer 2001, Poff et al. 1997, Roy et al. 2005, Walsh et al. 2005), many instream habitat restoration projects are designed to deflect high flows, reduce channel erosion incision, and provide structural complexity. The large gap between restoration goals and study design objectives often leaves project evaluations searching for the return of the biotic community (The Field of Dreams hypothesis – Palmer et al. 1997). However, this study demonstrated important relationships between fish response to stream channel enhancements and land use that may lead to better restoration project design, implementation, and management.

Fish responded to stream channel enhancement differently based on the land use category as well as the treatment applied. I accepted the hypothesis that fish in rural streams would select SH or SHWD more than LWD. Rural streams generally have greater habitat complexity and natural LWD contribution from riparian zones, and therefore addition of a few more logs would not likely stimulate a positive response from resident fish. An expected response was displayed by fish to just SH, choosing it about 50% of the time. However, the addition of both SHWD (over just LWD) indicated greater habitat use by rural fish, inducing a selection response.

Similarly, I hypothesized that fish in suburban streams would respond better to a combination of large woody debris and shade (SHWD) than other types of enhancement. Results from these experiments revealed that suburban populations showed an elevated response to SHWD compared to just SH and LWD. This result indicates that fish may either have a habitat element preference or that one of those components is not in great enough abundance to instill habitat selection. In rural streams, the combination of both SHWD may have created a synergistic habitat complexity that attracted fish to the treatment section over the control section. This relationship may also apply to suburban streams, where SH was not as enticing to fish as the complexity of both SHWD.

On the other hand, urban stream fish generally selected habitat enhancements that included shade more than the LWD treatment without. I originally hypothesized that urban fish would not select any one enhancement more than another due to the lack of available complex habitat and thus preference in urban channels. The greatest response in urban streams was found when overhead cover (SH) was added to the stream channel, however the response to SHWD was very similar. This is not surprising due to the impacts of the urban stream syndrome on the riparian canopy. Flashy stormflows not only incise channels and move instream LWD downstream, but erode urban streams to wider widths than small rural streams (Hammer 1972, Trimble 1997, Walsh et al. 2005). The riparian canopy provides less overhead cover to wider channels, and flow-induced erosion eliminates the development of undercut banks for urban fish populations. Thus, when provided with protection through the form of overhead shading, urban fish populations actively selected the enhanced habitat.

Patterns in fish species richness and abundance across the urban-rural gradient in this study were similar to other urban fish studies and reflect a shift in the species assemblage as well as the tolerance complex within the assemblage (Scott 2006, Morgan and Cushman 2005, Roy et al. 2005, Walters et al. 2003). The decrease in fish species richness in urban streams, coincident with a similar abundance to rural streams, indicates that urban fish assemblages are dominated by tolerant species (BND and CKB) who have taken the place of a more diverse intolerant species complex. These data agree with Morgan and Cushman (2005) in Maryland's eastern Piedmont fishes as well as other studies of changes in fish assemblages due to urbanization impacts (Scott 2006, Paul and Meyer 2001, Roth et al. 1996, Roy et al. 2005).

Recent examination of stream fish along the urban-rural gradient has revealed that the suite of urban stream symptoms may be influencing maturity and size. Fraker et al. (2002) found that urban blacknose dace experienced increased growth rates during their first year of life when compared to dace in rural streams. Yet in heavily urbanized watersheds (>90% urban land use), blacknose dace were smaller and younger at maturity due to a greater percentage of the population mature at age one (Fraker et al. 2002). In this study, both BND and CKB were smaller in urban streams than in suburban and rural streams. These differences may result in the creation of subpopulations due to environmental regulation of ecosystem structure and function.

Fish size presented some interesting relationships with habitat selection responses. Larger CKB were found in the treatment section than in the control, suggesting that individuals may compete for enhanced stream habitat. There is evidence that stream fish occupy different habitat niches at different stages (and thus size) of life. Size-dependent habitat segregation has been documented in banded sculpin *Cottus carolinae* (Koczaja et al. 2005), 'bullhead' *Cottus gobio* (Davey et al. 2005), and longnose dace *Rhinicthys cataractae* (Mullen and Burton 1998). In manipulative experiments, juveniles decreased their use of sheltered habitat (Mullen and Burton 1998) and selected shallow water (Koczaja et al. 2005) in the presence of adults. Given the choice of enhanced stream habitat in this study, smaller, juvenile CKB may have selected the less complex habitat in the presence of larger, adult CKB as a result of intraspecific competition. This behavior may also reflect reduced vulnerability to piscivorous predation. CKB have been known to cause prey fish species to move towards structurally simple pool habitat (Schlosser 1988). Conversely, adult CKB may have displaced the juveniles from prime habitat as a result of a dominance hierarchy, such as in salmonid feeding stations (Nakano 1995). In support of this, Bult et al. (1999) found that juvenile Atlantic salmon shifted habitats as a function of population density. Therefore, habitat use and selection may differ between life stages and be regulated by a variety of ecological interactions.

Some water quality results across the urban-rural gradient in this study were unexpected. I found the lack of differences among stream types in conductivity and dissolved oxygen somewhat surprising. The relationship between stream temperature and land use was not surprising. Recent studies have revealed higher stream temperatures with increased urban land use in the upstream watershed (Brasher 2003, LeBlanc et al. 1997, Wang et al. 2003). Warmer stream water may be acting in concert with other environmental stressors to reduce growth and size of fish populations in urban streams. BND and CKB are both considered coolwater stream fishes; however, they have been found at sites where temperature maximums have been exceeded (Chapter V, Wehrly et al. 2003). Therefore, peaks in stream temperature may not have an immediate impact on fish, yet may induce a biological adaptation within their life cycle to persist in urban ecosystems.

Finally, morphological differences in stream channels were not different across land use categories. Although some have suggested that rural forested channels are wider and follow a more natural meandering than deforested channels (Sweeney et al. 2005), other research has shown that urban channels are significantly wider due to intense erosion (Chapter 5, Hammer 1972, Trimble 1997, Bledsoe and Watson 2001). Erosion and a high density of pipes draining into the stream generates increased pool depth is considered a component of the urban stream syndrome (Walsh et al. 2005). The discrepancy in channel dimension differences across land use categories does provide a basis for good comparison though, since habitat features such as width and depth could not be associated with patch selection.

Urban stream environments are homogenized ecosystems due to the high intensity and frequency of disturbance, both on land and water (McKinney 2006). Channel and riparian habitat as well as the biotic community are simplified and subject to invasion by non-native species, thus entering a state of disequilibrium (Booth 2005, Scott 2006, Roy et al. 2005, Walters et al. 2003). Small stream fish assemblages in Maryland have low species richness but high abundance, composed mostly of cyprinids (Morgan and Cushman 2005), which is why this study focused on BND and CKB. This level of biotic homogenization is problematic when restoration of the stream community is expected to reach predisturbance conditions. BND and CKB are both pool dwelling species, so restoration projects in severely urbanized streams must consider that these species will likely be the first responders to instream habitat improvements, prior to a more diverse fish assemblage.

Based on the conclusions presented above, it appears that urban BND and CKB respond very well to overhead shading, providing protective cover over the stream channel. Measurements of stream water temperature indicated that urban systems are warmer than suburban and rural streams. Johnson (2004) performed a stream shading experiment and found that maximum water temperatures were significantly lower in stream segments shaded by black plastic sheeting. Combined, these lines of evidence imply that fish may actively seek out cooler patches within a stream reach, especially during the hottest part of the day. Since I conducted the patch experiment from about 10am to 4pm in open urban channels, the mechanism behind fish habitat selection may have been dominated by temperature or a response to both cooler temperatures and overhead cover.

Riparian management and instream habitat improvements are two approaches that can provide overhead cover and reduction in stream water temperature. Some stream channels have been enhanced with bank stabilization techniques using rootwads and whole logs to deflect flow away from the banks. The area around the logs, in turn, becomes promising habitat for fish commonly found in pools and runs by providing cover. Interestingly, urban fish did not select habitat with large LWD, which is potentially related to the homogenized simple channel effect. LWD and debris jams that are commonly seen in less disturbed systems provide not only protective cover, but food resources as well. Pools created around lodged LWD and smaller debris jams are also prime nesting sites. This is extremely important to remember and include in post-restoration monitoring plans.

An important aspect of stream restoration efforts is to make all attempts to remove the major source of stress in the system. In many urban streams, particularly in Maryland, this is an altered flow regime (Booth 2005, Poff et al. 1997, Roy et al. 2005, Walsh et al. 2005). If a stream still experiences flashy, high storm flows after bank stabilization or other instream habitat improvement, the restoration will fail, both physically and biologically (Booth 2005). Flow modification must be part of major habitat restoration efforts if successful biotic establishment is expected. In addition, reach-scale enhancements may not be enough to supply adequate habitat to reduce competitive and density-dependent exclusion from the restored reach. A greater ecological response requires longer stream sections to allow for movement and habitat selection (Moerke et al. 2004a).

In conclusion, this study has found that habitat patch selection and use by BND and CKB does vary among streams with different upstream land use. It is evident that based on what the fish assemblage is adapted to, whether it be moderate habitat complexity or moderate disturbance, the same species will not respond to instream habitat enhancements similarly. It is very important to understand not only what species exist within the stream network prior to the construction, but how they use the habitat that is currently available to them.

Secondly, urban systems do exhibit great variability in ecosystem structure and function (Meyer et al. 2005, Walsh et al. 2005). Fish actively selected habitat in urban streams that were typically shaded, providing protective cover in wide, simple, exposed channels. Although the response was greater when only shade was provided, urban fish also chose the combined treatment at a similar frequency. Thus, increasing the amount of overhead cover in urban stream channels would be beneficial for fish populations when implementing stream restoration practices.

In light of the numerous stream restorations without biological monitoring, it is understandable why stream channels are engineered for specific, local-scale physical results. However, due to the cost of these efforts, it is reasonable to incorporate a broader restoration scheme by challenging local and regional managers to tackle problems at a larger watershed scale, while keeping stream structure and function in mind. It is my hope that studies like this will provide insight to not only technical issues related to the restoration practices, but also to the enlighten those concerned about the biological component of stream ecosystems.

Tables

Table 1. Stream sites and accessory information used to survey habitat complexity in the eastern Piedmont of Maryland. Site names were derived from the original MBSS site name, but reflect the year of sampling. Latitude and longitude are presented in decimal degrees. County abbreviations are BA = Baltimore, BC = Baltimore City, HA = Harford, HO = Howard, MO = Montgomery, and PG = Prince George's. The ULU (urban land use) and UCat (urban category) represent the percentage of urban land use in the upstream watershed. The river basins represented in this study were ANA = Anacostia, BUS = Bush, GUN = Gunpowder, PAT = Patapsco, PAX = Patuxent, and POT = Potomac. Watershed area upstream from each site is represented in hectares.

Site	Latitude	Longitude	County	ULU	UCat	Basin	Area
BYNU-105-2005	39.3388	-76.2017	HA	0.00	RURAL	BUS	43
MPAX-107-2005	39.1166	-76.5772	HO	0.00	RURAL	PAX	130
LOCH-112-2005	39.5250	-76.7907	BA	0.00	RURAL	GUN	287
LOCH-114-2004	39.4948	-76.6847	BA	0.01	RURAL	GUN	631
GWYN-102-2005	39.4062	-76.8241	BA	0.13	RURAL	PAT	69
RKGR-119-2004	39.1685	-76.9720	HO	0.41	RURAL	PAX	298
SBPA-108-2004	39.3479	-76.9166	HO	0.49	RURAL	PAT	595

Site	Latitude	Longitude	County	ULU	UCat	Basin	Area
GWYN-112-2005	39.3955	-76.8114	BA	2.4	RURAL	PAT	92
LOGU-106-2005	39.4499	-76.4533	BA	2.5	RURAL	GUN	301
GWYN-105-2005	39.3888	-76.7709	BA	3.3	RURAL	PAT	499
LIGU-102-2005	39.5067	-76.4293	HA	6.9	RURAL	GUN	424
SENE-114-2005	39.2600	-77.2120	MO	13.1	RURAL	POT	277
PATL-103-2004	39.1919	-76.7421	HO	27.3	SUBURBAN	PAT	908
ANAC-110-2005	39.0953	-76.9275	MO	27.8	SUBURBAN	PAT	171
LWIN-104-2005	39.4752	-76.3752	HA	29.6	SUBURBAN	BUS	78
LIGU-105-2005	39.4721	-76.3874	HA	31.7	SUBURBAN	GUN	74
BA-119-2005	39.2660	-76.7920	BA	34.4	SUBURBAN	PAT	211
LOCH-123-2005	39.4283	-76.5810	BA	35.6	SUBURBAN	GUN	218
HO-104-2005	39.1560	-76.8190	HO	38.1	SUBURBAN	PAX	191
JONE-109-2004	39.4067	-76.7280	BA	41.2	SUBURBAN	PAT	306
LPAX-116-2004	39.1872	-76.8614	HO	41.9	SUBURBAN	PAX	485
HO-120-2004	39.2740	-76.8410	HO	42.5	SUBURBAN	PAX	242
LIBE-107-2005	39.5739	-76.9867	CA	43.7	SUBURBAN	PAT	143
GWYN-107-2005	39.4572	-76.8018	BA	45.8	SUBURBAN	PAT	605
PATL-116-2005	39.2600	-76.7660	HO	61.4	URBAN	PAT	164

Site	Latitude	Longitude	County	ULU	UCat	Basin	Area
ANAC-116-2005	39.0226	-77.0307	MO	62.6	URBAN	ANA	906
LOGU-103-2004	39.4043	-76.5107	BA	64.2	URBAN	GUN	267
PATL-194-2005	39.1416	-76.4365	BA	62.5	URBAN	PAT	794
MO-127-2004	39.0960	-77.0130	MO	64.3	URBAN	POT	101
BACK-113-2004	39.3667	-76.5229	BA	64.9	URBAN	PAT	347
BA-126-2004	39.2170	-76.4571	BA	66.6	URBAN	PAT	854
PAXU-105-2005	39.1042	-76.8884	PG	69.1	URBAN	PAX	95
CABJ-102-2005	39.0714	-77.1518	MO	73.0	URBAN	ANA	238
BA-128-2004	39.3420	-76.5140	BA	73.9	URBAN	PAT	387
LOGU-190-2005	39.2413	-76.3448	BA	74.9	URBAN	GUN	140
MO-126-2004	39.0710	-77.080	MO	80.8	URBAN	POT	202

Table 2. Individual species responses to experimental treatment effects. BND and CKB data were tested separately to determine whether one species was responsible for treatment effects in the combined analysis. No significant effects were found for either species (*BND = blacknose dace; CKB = creek chub*).

	Effects	**df**	**F-value**	**P-value**
BND				
	Treatment	2, 27	2.3	0.12
	Land	2, 27	0.9	0.44
	Treatment*Land	4, 27	1.2	0.34
CKB	**Effects**	**df**	**F-value**	**P-value**
	Treatment	2, 27	1.2	0.33
	Land	2, 27	1.6	0.22
	Treatment*Land	4, 27	1.5	0.24

Table 3. Water quality and discharge values for rural, suburban, and urban streams in 2004 and 2005. Temperature (°C), conductivity (mS/cm), dissolved oxygen (DO; mg/L), pH were measured at each site. Discharge (m^3/s) was calculated from depth (m), lateral location (m) and water velocity (m/s) measurements. Temperature was significantly ($P < 0.01$) different among land use categories [*ULU = urban land use category; SEM = standard error of the mean*].

Parameter	ULU	Mean	SEM
Temperature			
	Rural	18.84	0.55
	Suburban	19.13	0.43
	Urban	21.19	0.62
Conductivity	Rural	0.305	0.047
	Suburban	0.398	0.046
	Urban	0.460	0.049
DO	Rural	8.47	0.27
	Suburban	8.05	0.28
	Urban	7.38	0.39
pH	Rural	7.37	0.14
	Suburban	7.12	0.09
	Urban	7.34	0.12
Discharge	Rural	0.017	0.005
	Suburban	0.010	0.003
	Urban	0.006	0.002

Table 4. Analysis of stream channel morphology at rural, suburban, and urban stream sites. Maximum depth (m) was measured only in 2005, however channel width was measured through the 2004-5 sampling season. Average width (m) is the average of channel width at 0m, 10m, and 20m in the experimental reach, where 10m is the boundary between the control and treatment sections [*ULU = urban land use category; SEM = standard error of the mean*].

Parameter	ULU	N	Mean	SEM
Max Control Depth (m)	Rural	7	0.29	0.048
	Suburban	8	0.31	0.033
	Urban	8	0.42	0.051
Max Treatment Depth (m)	Rural	7	0.34	0.046
	Suburban	8	0.40	0.110
	Urban	8	0.38	0.049
Average Width (m)	Rural	12	2.74	0.315
	Suburban	12	2.72	0.279
	Urban	12	3.59	0.403

Figures

Figure 1. Map of habitat patch stream sites found in the Bush, Gunpowder, Patapsco, Patuxent, and Metro region of the Potomac River watersheds. Rural (< 15% ULU), suburban (27-46% ULU), and urban (> 60% ULU) sites are differentiated by color.

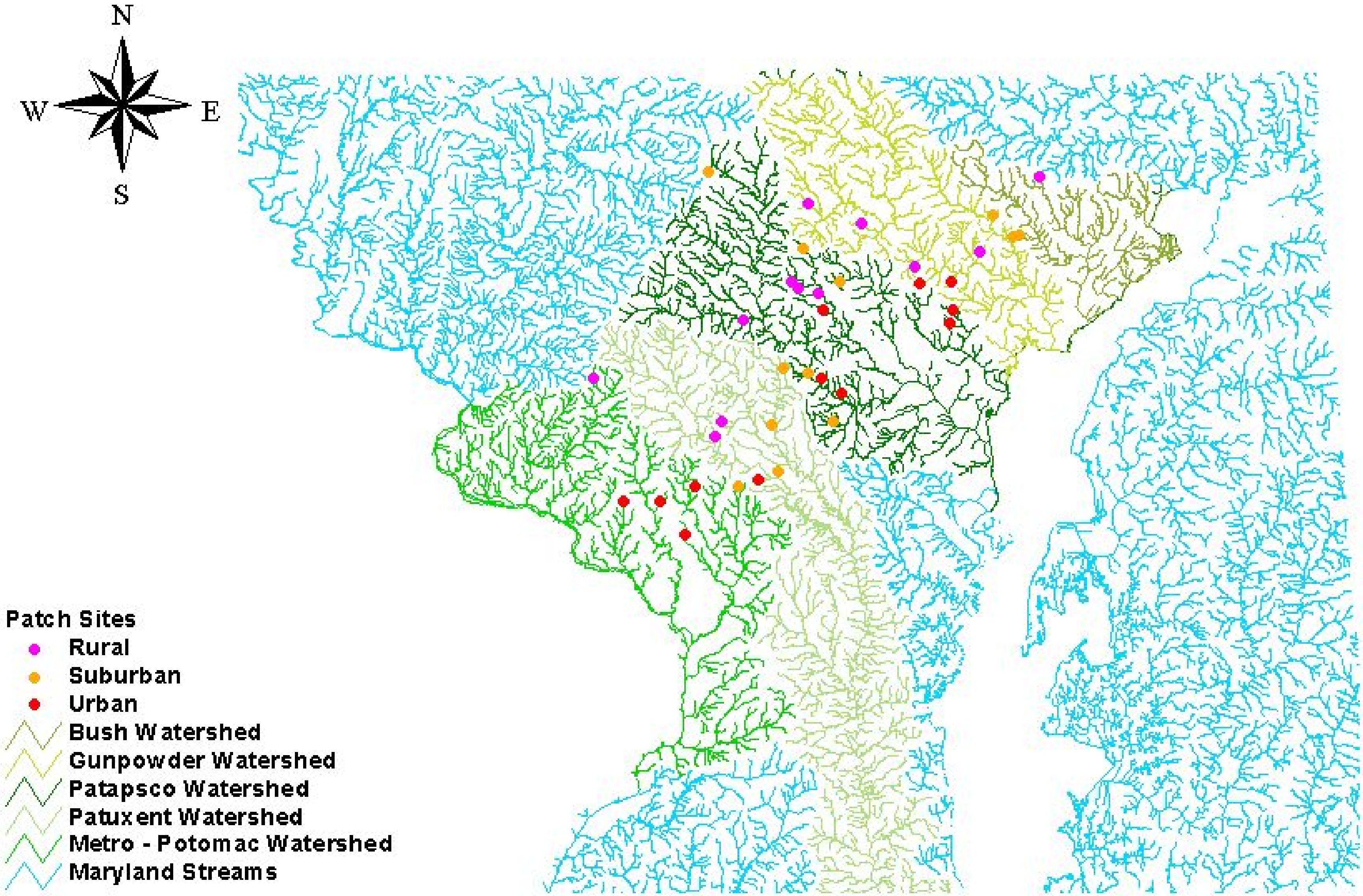

N
W
E
S
Patch Sites
Rural
Suburban
Urban
Bush Watershed
Gunpowder Watershed
Patapsco Watershed
Patuxent Watershed
Metro - Potomac Watershed
Maryland Streams

Figure 2. Diagram of the habitat patch experiment. The position of the treatment and control sections were randomly chosen prior to arrival at the site.

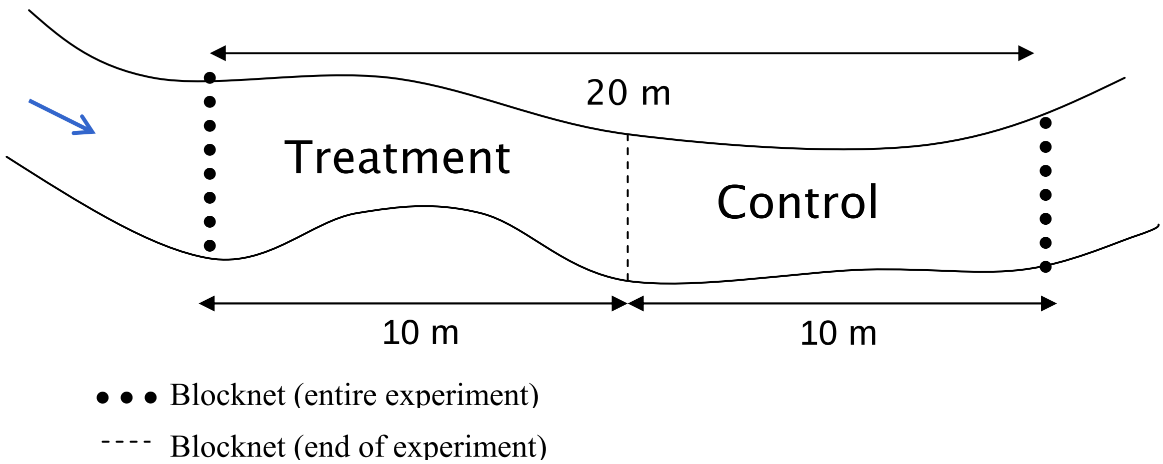


Figure 3. Species richness at rural, suburban, and urban stream sites at the beginning of the experiment. Richness of rural stream assemblages is significantly higher than richness of urban streams (* $P < 0.05$). Bars represent mean ± SEM. Letters indicate homogeneous groups.

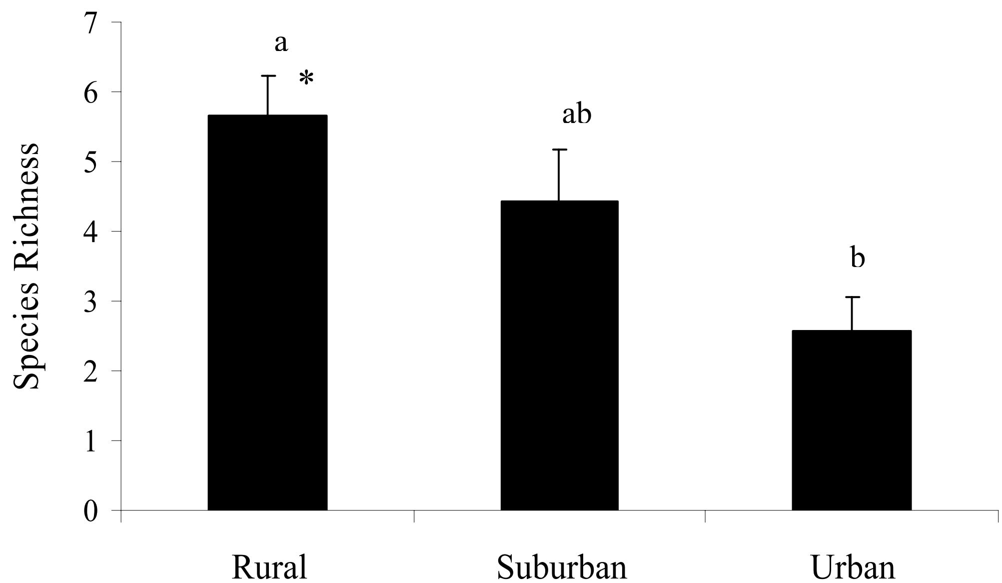


Figure 4. Relative abundance of fish found in the 20 m segment at the beginning of the experiment in rural, suburban, and urban streams. There was no significant difference in abundance across ULU categories. Bars represent mean ± SEM.

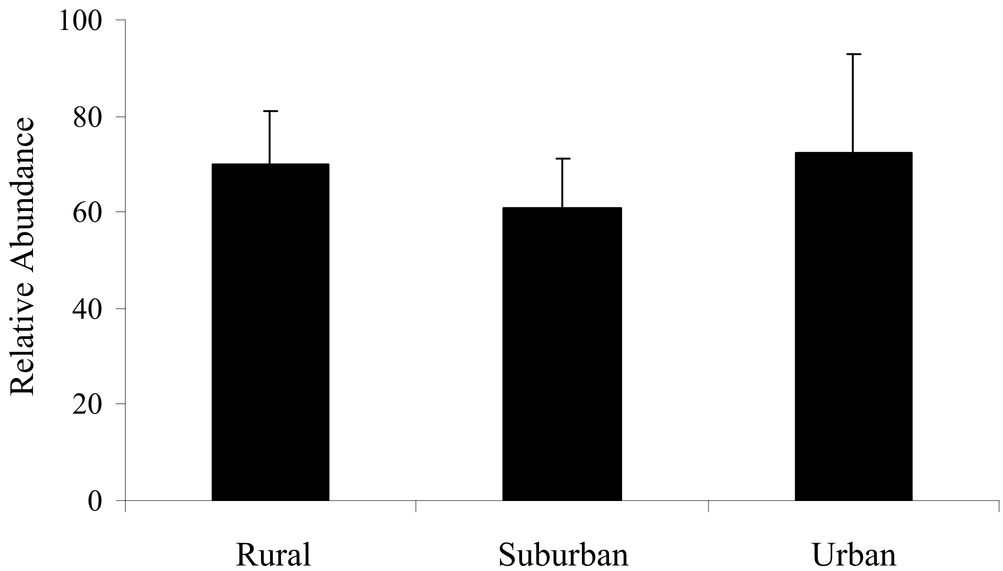


Figure 5. Fish response to stream channel enhancements in rural, suburban, and urban streams. One of three enhancements (woody debris, shade or both woody debris and shade) were paired with a control (no enhancement) to each site in the study. Treatment proportion represents treatment effects and equals the number of fish collected in the treatment over the total number of fish found in the 20 m segment at the end of the experiment. Bars represent the mean ± SEM.

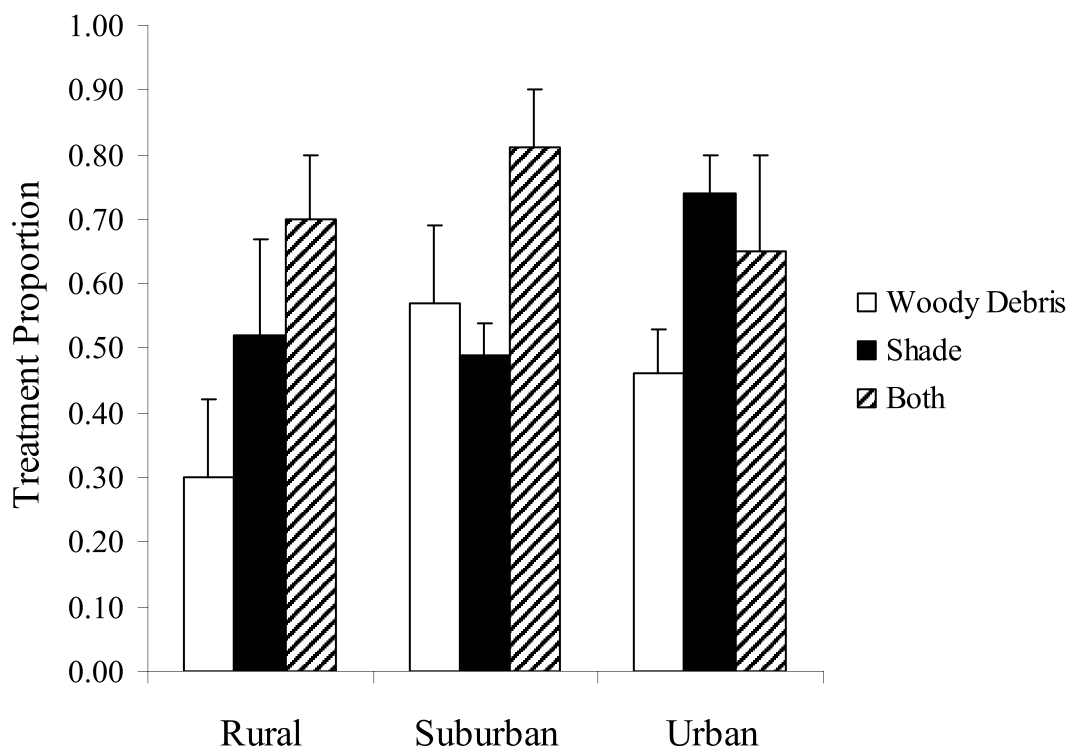
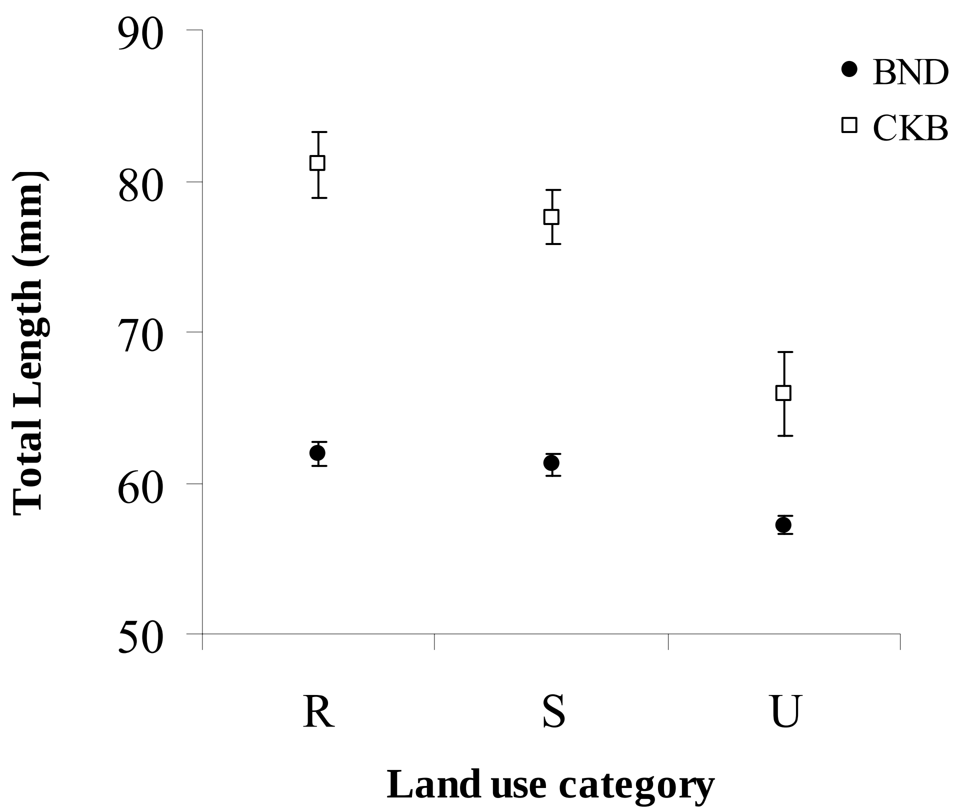


Figure 6. Mean total length (±SEM) of BND and CKB in urban, suburban, and rural stream populations. Urban BND and CKB were significantly smaller than both suburban and rural fish ($P << 0.001$ and $P < 0.01$, respectively) however there was no difference in suburban and rural BND and CKB length. [*U = urban; S = suburban; R = rural; BND = blacknose dace, CKB = creek chub*].



Chapter 5: Movement patterns of stream cyprinid subpopulations relative to habitat in urban and rural watersheds

Abstract

The ecological explanation for both resident and mobile fish within a population has been long debated, with some fish exhibiting restricted movement while others use widespread stream habitat. Few studies have questioned how environmental quality, particularly urbanization, affects ecological interactions and population dynamics of stream biota. Fish movement patterns were hypothesized to differ in streams populations from urban and rural watersheds, specifically differing in the proportion of "movers" and "stayers" and size of homerange. Two cyprinids, blacknose dace (BND) *Rhinichthys atratulus* and creek chub (CKB) *Semotilus atromaculatus* in eight streams (four urban, four rural) were individually marked with visible implant elastomers and their location was monitored from July to October. The proportion of movers and stayers did not differ significantly between urban and rural streams, however urban fish display significantly ($P < 0.001$) greater home ranges than rural fish. The distribution of movement distances in rural populations was more leptokurtic than urban populations. Species-specific patterns were also evident in urban streams. Urban BND movers were significantly ($P < 0.01$) longer than stayers, and urban CKB grew less than rural CKB ($P < 0.05$). Urban CKB exhibited a positive relationship between length and distance moved (Adj-R^2 = 0.19, $P < 0.05$). Urbanization appears to increase competition within simplified fish assemblages, causing fish to diffuse throughout the stream reach. This research on fish

movement in contrasting stream environments proposes mechanisms behind movement of fish populations, and gives insight to fish ecology in degraded systems.

Introduction

The controversy over the explanation of temporal changes in fish abundance and distribution patterns dominates the literature on freshwater fish movement (Gerking 1953, 1959, Linfield 1985, Rodriguez 2002, Gowan and Fausch 2002, Hilderbrand and Kershner 2000, Hill and Grossman 1987a, Smithson and Johnston 1999, Gowan et al. 1994, Larson et al. 2002). Although much of the movement literature examines salmonid species, recent studies have described patterns of fish movement and home range in non-salmonid species (Gilliam and Fraser 2001, Goforth and Foltz 1998, Petty and Grossman 2004, Skalski and Gilliam 2000, Smithson and Johnston 1999, Larson et al. 2002, Lonzarich et al. 2000). The restricted movement paradigm, as defined by Gowan et al. (1994), declares that fish display restricted, sedentary lifestyles, residing in the same pool or stream reach for their entire life (Gerking 1953, 1959). Alternatively, others suggest that fishes use and move through large expanses of stream networks, residing in multiple habitat patches over time (Linfield 1985, Gowan et al. 1994). The debate between these two hypotheses involves explanations of population dynamics with both sedentary and mobile individuals. Fish residing in a small region of a stream or river over long periods of time have been referred to as "stayers" or the resident subpopulation, while those that continually explore new habitat have been called the "movers" or the mobile subpopulation (Funk 1957, Hilderbrand and Kershner 2000, Skalski and Gilliam 2000, Colyer et al. 2005, Larson et al. 2002). This life history diversity within a species challenges scientists ability to make accurate estimates of population size and structure, thereby restricting our ability to promote reasonable conservation practices, and test hypotheses of the effects of both abiotic and biotic influences on fish assemblages.

Fish habitat use varies greatly by species. Species that reside in the water column of pools (limnetic) occupy a different niche than benthic species found in riffles, and therefore play different ecological roles within the ecosystem. Numerous studies of habitat use and movement between habitat patches indicate that benthic and limnetic fish species behave differently (Johnston 2000, Shaefer 2001, Freeman and Grossman 1993, Goforth and Foltz 1998, Gowan and Fausch 2002, Petty and Grossman 1996, Petty and Grossman 2004, Smithson and Johnston 1999, Hohausová et al. 2002, Thompson et al. 2001). Habitat use may fluctuate with habitat availability, life history phases, and interspecific interactions (Schlosser 1987), thus producing reasons for movement between habitat patches. The proposed mechanism for movement in many of these studies are ecological, however few have questioned if environmental quality determines these patterns (Hohausová et al. 2002).

Urban land development has spread at dramatic rates throughout the nation, creating a serious threat to small stream networks (Paul and Meyer 2001). Newly urbanizing areas, suburban, and mature urban settings in metropolitan regions produce distinct stressors to stream health (Angermeier et al. 2004). Residential and commercial development encroaches on intermittent and perennial channels by narrowing the natural riparian buffer, modifying stream habitat structure, degrading water quality, and altering the natural flow regime by establishing stormwater sewers and drain pipes throughout the watershed (Angermeier et al. 2004, O'Neill et al. 1997, Paul and Meyer 2001, Poff et al. 1997).

Impervious surface cover has become an environmental indicator for freshwater ecosystems due to their ability to deliver overland runoff to streams during precipitation events (Arnold and Gibbons 1996, Walsh et al. 2005a). Runoff commonly gets routed away from buildings and roads through stormwater drain pipes, directly into stream channels thus eliminating the absorption and percolation of precipitation through soil to recharge groundwater. Overland flow collects pollutants and sediment from both impervious and porous surfaces, thus delivering degraded water quality to the stream channel. In addition to polluting stream water, intense volumes of runoff reach the stream channel and travels down the stream network faster than in rural ecosystems.

This altered flow regime becomes a major source of instream habitat and channel morphology degradation, including heavy erosion of streambanks, downcutting, reorganization of regular channel sub-unit sequences, and decreased habitat structure and complexity (Paul and Meyer 2001, Poff et al. 1997, Walsh et al. 2005b). Stream channel reaches become homogenized such that the length of each habitat unit, as well as the spacing or interval length between habitat units may be longer in urban streams compared to streams in forested watersheds. In addition, channel widening may create shallow, high-risk areas (e.g. riffles during low baseflow) through which fish may be reluctant to move. Lonzarich et al. (2000) indicated that long stretches of riffle habitat between pools restricts movement of pool species. These pervasive modifications to instream fish habitat have been shown to drastically impact biotic communities. Fish assemblages, in particular, have been shown to respond to urbanization effects by decreased species richness and abundance of sensitive species, with negative impacts on growth rate, maturation and recruitment (Pirhalla 2004, Limburg and Schmidt 1990, Morgan and

Cushman 2005, Paul and Meyer 2001, Tabit and Johnson 2002, Weaver and Garman 1994, Fraker et al. 2002, Wang et al. 2003).

Increased study of urban stream ecosystems has led to the conception of the "urban stream syndrome" (Meyer et al. 2005), which is defined by a set of characteristics that describe the ecological degradation of ecosystem patterns and processes in urban watersheds (Groffman et al. 2005, Walsh et al. 2005b). Although the major symptoms urban streams exhibit have been described (flashy hydrograph, elevated nutrient and contaminant concentrations, altered channel morphology and stability, and reduced species richness; Walsh et al. 2005b) many aspects of urban stream ecosystems are poorly studied. In particular, how urbanization affects ecological interactions and population dynamics of stream biota remains unknown.

Since land use change modifies the hydrologic regime, habitat complexity, and biotic communities, I question whether stream habitat quality relates to differences in fish movement patterns. First, I hypothesized that the proportion of movers and stayers would differ between rural and urban fish populations. Secondly, I hypothesized that fish in urban streams will demonstrate a larger home range than rural fish in pursuit of suitable habitat. These hypotheses suggest that fish habitat use and movement patterns differ depending on stream habitat, which relate directly to the effectiveness of ecological monitoring, population estimates, and to the overall understanding of stream fish ecology. To test these hypotheses, I marked individual fish and monitored their location within the stream reach to compare movement patterns in streams draining urban and rural watersheds.

Methods

Two stream cyprinids, *Rhinichthys atratulus* blacknose dace (BND) and *Semotilus atromaculatus* creek chub (CKB), were selected for this study due to their presence in both rural and urban watersheds. These species are considered pollution-tolerant and are commonly found in the most degraded urban streams, and thus serve as excellent study organisms for this type of comparison (Pirhalla 2004, Morgan and Cushman 2005, Boward et al. 1999). I designed a marking program to identify individual fish and conducted a mark-recapture experiment with BND and CKB in two urban and two rural streams per year for two years (n = 8).

Study sites

The Maryland Urban Fish (MUF) database contains the results of sampling of Maryland streams (from the Maryland Biological Stream Survey = MBSS) conducted in 1995-1997 (Round 1) and 2000-2004 (Round 2). The sampling program involved sampling of randomly chosen stream segments conducted by the Maryland Department of Natural Resources or The University of Maryland Center for Environmental Science. A comprehensive data record (physical habitat, water quality, fish collection, and land use characterization) was collected for each 75 m stream site sampled. A fish index of biotic integrity (FIBI; Roth et al. 1998, Roth et al. 2000), benthic index of biotic integrity (BIBI; Stribling et al. 1998) and physical habitat index (PHI; Hall et al. 1999, Paul et al. 2003) were calculated for each site within the database. The MUF database includes first, second, and third order streams in the eastern Piedmont (EP) and Coastal Plain physiographic provinces in Maryland, and included only those sites that had less than

65% agricultural land use in the upstream watershed and less than 8 mg/L dissolved organic carbon (e.g. blackwater).

Specific sites for the fish movement study were chosen from the MUF database where BND and CKB were previously collected in abundance and landuse characteristics matched my research objective. To compare fish movement in urban and rural watersheds, four rural and four urban EP streams were selected giving a total of eight stream sites. Two rural and two urban streams were assessed each sampling season for a total of two years (Table 1). A rural stream was described as a site with less than 15% urban land use in the upstream watershed, whereas an urban stream was described as a site with greater than 60% urban land use in the upstream watershed. All sites were first order tributaries to river basins located in the EP (Figure 1).

Benson Branch (HO; Howard County, Maryland) is a first order stream in the Middle Patuxent River basin (Table 1). The upstream watershed from the sampling site is 421 ha, of which 1% is urban land use. The remaining landuse in the watershed consists of 56% agriculture, 42% forest, and 1% water. Previous MBSS evaluation scored this site with both a FIBI and BIBI of 3.7 on a scale of 1 to 5, and a physical habitat index of 94 out of 100.

Keysers Run (LIBE; Baltimore County, Maryland) is a first order stream in the Liberty Reservoir watershed (Table 1). The upstream watershed consists of 59% agriculture, 35% forest, and 6% urban land use (total area = 161 ha). The FIBI and BIBI ratings for this stream were 4.1 and 4.3, respectively, while the PHI was 70. This stream, as well as HO, was classified as a rural stream since the percent urban land use was less than 15%.

Magruder Branch (SENE; Montgomery County, Maryland) is a first order stream in the Seneca Creek basin (Table 1). The watershed consisted of 13% urban land use, 60% agriculture, and 27% forested land (total area = 277 ha). The FIBI and BIBI were 4.1 and 2.6 respectively, while the PHI was 88. Thus, this stream was included in the rural stream category.

The fourth rural stream sampled in this study was an unnamed tributary to the Middle Patuxent River (MPAX; Howard County, Maryland). The land use in this stream's watershed (total area = 129 ha) is dominated by agriculture (59%) and forest (39%), with some water (2%) and no urban land use (0%). This stream did not display FIBI (3.0), BIBI (3.4), and PHI (36) values as high as other rural streams.

Jennifer Branch (LOGU; Baltimore County, Maryland) is a first order stream in the Lower Gunpowder River basin (Table 1). This stream was classified as an urban stream because its upstream watershed (267 ha) consisted of 64% urban land use, 27% forest, and 9% agriculture. This stream was sampled previously giving both FIBI and BIBI scores of 2.6 on a scale from 1 to 5, and a PHI score of 55 out of 100. About 20 m of the right streambank in the study [recapture] segment had been previously stabilized by a common restoration practice using large woody debris and tree stumps.

Stemmers Run (BACK; Baltimore County, Maryland) is a first order stream in the Back River basin (Table 1). Similar to LOGU, this site was composed of 65% urban land use, 21% forest, and 14% agriculture in its 347 ha watershed. However, its FIBI was rated at 1.9 and BIBI = 2.1 and the PHI was only 18. Due to the percent urban land use in the watershed, LOGU and BACK were both classified as urban streams.

Cabin John Creek (CABJ; Montgomery County, Maryland) is a first order stream flowing into the Potomac River (Table 1). This stream is the most urbanized of the study sites (73%), but also has 20% forest, 6% agriculture, and 1% water within its watershed (total area = 238 ha). The FIBI and BIBI were both low at 1.9 and 2.3, respectively, however the PHI was not calculated. This stream also exhibited about 50 m total of engineered bank and channel flow stabilization practices comprised of large woody debris, cobble, and boulders.

Sligo Creek (ANAC; Montgomery County, Maryland) is in the Anacostia River basin and is a first order stream (Table 1). Its watershed (total area = 367 ha) consists of 63% urban land use, 32% forest, and 5% agricultural land. Like CABJ, this stream was classified as urban for this study. Again, the PHI was not calculated, but the FIBI was rated at 1.2 and the BIBI was given a value of 2.3, both on a scale of 1 to 5. The study segment at Sligo Creek included about 5 m of cemented cobble along the right bank constructed to reduce erosion.

Habitat assessment

Although physical habitat was previously assessed at all sites using MBSS methods, a more detailed and comprehensive summer stream habitat assessment was completed prior to fish marking. This was done to ensure that any correlations between habitat and movement were as accurate as possible, since stream habitats are dynamic. At each site, a stream map was drawn for the entire sampling segment, including the relative position and lengths of channel subunits (pool, riffle, run, and glide), sinuosity, position of woody debris, rootwads, debris jams, tributaries, and bar formation. Estimated lengths and number of each channel unit were recorded. The number of every rootwad within 5 m of the streambank tallied, and identified to species to the lowest level possible. Woody debris ≥ 10 cm in diameter and 1 m in length and the number of debris jams (wedged piles of woody debris and other organic matter greater than 0.25 m^2) were also documented. The linear extent of eroded streambanks, maximum height of erosion, and linear length of bar formations were estimated and recorded for each 75 m segment.

Stream channel transect measurements were made at each flag (15 m intervals) to give a more comprehensive picture of the study site. Channel characteristics including stream width, thalweg depth, and thalweg velocity were recorded. The percent shading over the channel, and the type and extent of cover within a 10 m buffer of riparian vegetation, was described.

Water quality measurements were made above the 75 m sampling segment so as to not sample in disturbed water. A Hydrolab® Quanta® was placed in the middle of the stream channel to collect single recordings of stream water temperature (°C), pH, dissolved oxygen (mg/L), and conductivity (mS/cm) measurements.

Hydrologic sampling

A crest piezometer constructed of commercially available PVC pipe (5 cm diameter) was constructed with a 14 gauge wire perched inside and monitored at each stream site throughout the 2004 (July – October) and 2005 (June – October) sampling periods. Prior to installation, holes were drilled into the bottom third and one hole at the top of a 1.5 m pipe to allow stream water to enter the piezometer once positioned in the streambed. A naturally structured, protected position was chosen for the piezometer at each stream site to reduce the chance of washout during high stormflows. The pipe was anchored in the substrate using a rubber mallet and secured with large cobble around the base. A wire the length of the pipe above the stream sediment was cut, sanded to increase roughness, and coated with blue chalk. The water level height on the wire, as well as the total wire length was recorded. In case the pipe moved vertically or substrate height changed during stormflows, the length of PVC pipe above and below the water level was also recorded. A PVC cap was placed over the top of the pipe to protect the chalk level indicator from precipitation and debris.

Stream sites were visited periodically to record the water level of the piezometer. Measurements of wire water level, maximum water level (where blue chalk was visible), PVC water level, and PVC height above the water were recorded on each visit. These data were later used to estimate the maximum stage height between sampling periods.

Water quality and discharge measurements were taken at each site during piezometer and fish capture visits in 2005. Water quality and discharge were not sampled at sites in 2004 at times other than during fish capture. To measure discharge, a meter tape was stretched across the stream channel perpendicular to flow and secured, while depth and velocity measurements were taken at regular intervals across the channel. Depth was measured using a meter stick, to the nearest cm and velocity was measured using a Flow Mate flowmeter mounted on a wading rod and taken at 0.6 times the depth from the surface, which was recorded to the nearest hundredth m^2/s. Measurements were repeated at the same location each time the stream was visited, which was selected where stream flow was confined and minimally turbulent. Discharge was calculated in units of m^3/s, normalized to units of $m^3/s/ha$ for watershed area, and used for comparison of stream habitat and fish movement.

Four HOBO Water Temp Pro temperature loggers (Onset Computer Corporation, Massachusetts, USA) were installed in the 2005 stream sites to measure water temperature throughout the fish sampling period. They were installed in late June and retrieved in October. Loggers were programmed using BoxCar® Pro (v. 4.3) to collect temperature (ºC) on a 30-min interval. Each logger was fastened securely with multiple zip ties to submerged woody roots in the 75 m marking segment at both urban and rural sites. Data were downloaded into BoxCar® Pro and subsequently exported into Microsoft Excel for graphical presentation.

Fish collection and marking

A 75 m experimental segment bounded by riffles was selected at each site where fish collection and marking occurred in late July 2004 and in mid-June 2005. Blocknets were installed at the upstream and downstream ends of the 75 m segment, secured with cobble along the stream bottom and stakes along the streambank. Boundaries of each pool-glide subunit within the 75 m segment were defined and 19-liter buckets were placed along the streambank at each riffle.

Fish were collected by electrofishing (Smith-Root® model 12 backpack battery electrofisher). The electrofisher voltage was set to the lowest possible setting that would successfully stun, but not injure, any fish. The entire segment was then sampled using single-pass electrofishing to reduce exposure and mortality with multiple recapture collections. All captured fish were placed into water filled buckets. As sampling progressed upstream, fish were placed into separate buckets representing distinct habitat units within the segment. These habitat units were composed mostly of pools and some run channel subunits, although they were collectively referred to as the “home pool” between riffles.

All fishes were identified to species, counted, and recorded streamside. BND and CKB were held in the buckets in which they were collected so that each fish could be anesthetized and marked later. Every BND and CKB captured within the 75 m sampling segment that was ≥ 40 mm in total length was included in this study. All other fish species were processed and released downstream of the lower blocknet.

To identify individual fish upon recapture, I used visible implant elastomer tags (VIE) made by Northwest Marine Technologies Inc., Washington because of the color selection, marking technique, and flexibility in creating a unique marking protocol. BND, the smaller of the two cyprinids, range from 40 to 70 mm standard length (Jenkins and Burkhead 1993) which presented challenges in using most of the other standard marking technologies such as Floy tags and passive integrated transponder (PIT) tags. VIEs were injected subcutaneously in different anatomical locations with different colored elastomers (red, orange, yellow, and green). Using a combination of six anatomical positions, four elastomer colors, and two marking locations, 244 individual fish could be marked uniquely at each stream site. In 2004, I used the following anatomical marking positions: left cheek, right cheek, pelvic (girdle area), left caudal peduncle, right caudal peduncle, and caudal (dorsal side of caudal fin insertion). Based on assessment of anatomical mark frequency from recapture in 2004, I changed some of the positions in 2005 in order to recapture more fish with high quality marks. Instead of using cheeks, I added left and right anal (ventral side of the anal fin insertion), and left and right pectoral (ventral-posterior to pectoral fin insertion).

Fish were anesthetized in a 50 mg/L MS-222 solution. Fish were retrieved from home pool buckets with a small aquarium net and were placed two at a time into the anesthetic solution. Once fish displayed a loss of equilibrium and control of their position in the water column, they were removed from the solution and measured for total length. Larger fish, especially CKB, required more time and a stronger anesthetic solution (~75 mg/L).

Fish were marked by inserting the needle subcutaneously and releasing the elastomer as the needle was drawn out of the space created in the tissue. Excess elastomer was wiped off the needle tip on a damp sponge. Marking location combinations were determined and recorded prior to marking so that the procedure could be performed expeditiously. Each fish received a single injection in two different positions and was immediately placed into a recovery bucket filled with aerated stream water and then in a live well placed in the stream. Fish were released into the same pool that they were originally captured from after recovery from marking.

Recapture

In 2004, fish were recaptured twice, once in late August, and once in early October. I recaptured fish three times in the 2005 sampling season in July, August, and October. Fish were recaptured in October to determine if seasonal changes resulted in different movement patterns. A recapture segment of 275 m in length was sampled, adding a 100 m upstream and 100 m downstream segment to the original 75 m marking segment. In 2005, I sampled an extra 50 m on each end of the lower and upper 100 m recapture segments (total of 375 m) to determine if fish that were not captured in the original 275 m segment were just outside. This was done at one rural (SENE) and one urban (CABJ) stream.

The marking segment (bounded by blocknets) was resampled using single-pass electrofishing, and the upper and lower recapture segments were subsequently electrofished to increase the frequency of fish recapture as well as to detect extended distances traveled. Once fish capture from each segment was completed, individuals were identified and pool residency was noted. Mark positions and colors, total length, and any observation of note were recorded for all BND and CKB, and all fish were released into the same pool from which they were removed. In July and August 2005, I marked all unmarked blacknose dace and creek chub captured in the 75 m marking segment to increase the sample size of the study. In addition, if fish were found with only one visible mark, they were remarked with a new position/color combination that was not previously used.

To determine the recapture pattern for each stream, the original marking data including the anatomical locations marked, elastomer color used, and total fish length was tabulated. Fish that were recaptured once, twice, or three times were then matched to a fish dataset based on the observed mark locations, colors, and total length. If only one mark was observed upon recapture, an attempt was made to match the data from recapture to the initial marking data using fish length and pool residency. Recapture efficiency for fish with two visible marks was calculated by dividing the number of fish found upon recapture by the total number of fish. This percentage was calculated for each species in each stream category.

My study addressed the movement of fish, primarily based on pool residency differences between the initial marking date and recapture dates. I determined whether fish were movers or stayers depending on the pool in which they were found upon recapture. Fish that were found in a different pool upon recapture from the pool in which they were collected, marked, and released were considered "movers". If an individual was found multiple times in one pool but found once in a different pool it was still considered a mover. Fish that were only recaptured in the same pool in which they were originally collected and marked were defined as "stayers". All individuals that were classified as "movers" were assessed for distance traveled between capture dates as well as the direction of movement (upstream or downstream).

Total unsigned (without direction) movement was defined as the cumulative distance traveled between the collection dates. For example, if an individual was recaptured each time the stream was sampled, the total signed movement was equal to the distance between the marking location and recapture location #1, plus the distance traveled between recapture location #1 and recapture location #2. If an individual was only collected two times, the distance traveled was the total distance between those dates. Signed movement (includes directionality) was also calculated, providing movement patterns of individuals who moved in one direction (upstream or downstream) between two dates and then were collected in a pool in the opposite stream direction upon subsequent recapture. All distance measurements were calculated using midpoints of the home pool in the 75 m marking segment and the capture pools in the any of the 275 m of the recapture segment. Pool length was measured to the nearest 0.5 m in all pools/runs in the 275 m segment; maximum pool depth and pool width of those in the 75 m marking segment were also measured at each stream site in order to describe the characteristics of the pools fish selected to occupy.

Calculations and statistical analyses

ANOVA and t-tests were used to conduct analyses on all habitat variables, discharge, and water quality parameters. Fish that were recaptured provided various aspects of data. Population abundance was calculated for each site. Estimates of distance moved was used calculate home range, and estimates of length were combined with intercapture period data to estimate growth. I also assessed the relationship between movement status, growth, and environmental variables.

As capture-recapture data for multiple dates at each sampling site were available, I used the Jolly-Seber open population model to estimate population abundance for both BND and CKB. The Jolly-Seber model equates abundance (N) to the number of fish recaptured with and without marks (n_i) multiplied by the number of marked fish in the catch (M_i) all divided by the number of fish marked prior to sampling [(m_i); Pine et al. 2003).

$$N = (n_i M_i)/(m_i)$$

It is assumed that catchability remained constant during all fish captures and that marked fish not caught emigrated out of the segment or died. In 2004 I recaptured fish twice, therefore producing two estimates of population abundance. In 2005, I recaptured fish three times which allowed three different estimates of the population. Calculated population abundances were averaged at each site across sampling periods.

Marked fish and recapture data were compared for differences in site, stream type and year using ANOVA. Differences between mover and stayer proportions were analyzed using likelihood ratio Chi-square statistics (G-test; Gotelli and Ellison 2004). Distance moved was not normally distributed and could not be transformed due to a high frequency of zeros; therefore, it was analyzed using the Kruskal-Wallis test (Sokal and Rohlf 1981).

Home range estimates were calculated by estimating the distance between the most upstream and most downstream pool where a marked fish was released and recaptured, plus the length of each pool. Since distance was calculated using midpoints, I added half of the pool length (on each end) to the distance traveled to include the habitat in each pool. ANOVA was used to assess differences between urban and rural home range, as well as differences between species.

I evaluated the movement data on a monthly basis to determine if fish dispersal was dependent on the time of year or monthly conditions. The data were stratified based on the month of recapture, using the distance moved from last capture regardless of whether it was a month or more before. If a fish was captured in July, August, and October, the longest distance traveled (July – August or August – October) was used for analysis, while the other distance value was discarded. This was done to maintain independence among data points since the same fish could produce more than one data value over the entire sampling period. At rural sites, four values from August and five values from October were removed, while in the urban data, five values in July, five values in August, and 11 values in October were removed. Since these data included a large number of zeros and were not normally distributed, I conducted a Kruskal-Wallis non-parametric analysis on each dataset (Sokal and Rohlf 1981). Finally, to determine if stormflow or maximum stage height was the mechanism behind BND and CKB movement patterns, I used linear regression of monthly movement on the difference between maximum and baseflow stage height.

Length and growth rates of BND and CKB were assessed for their relationship with distance traveled. I used an ANOVA to test the *a posteriori* hypothesis that lengths and growth rates between movers and stayers were similar. When mover and stayer variance was similar, I removed the stayers from the dataset since the associated distance traveled was "0" and thus could not be transformed or otherwise analyzed using parametric tests. Length at time of marking was used for the linear regression analysis between length and distance. I regressed growth rate on distance by species and stream type to determine if a causal relationship occurred. I assumed that growth was constant over the summer (June – August) season, and over the fall (August – October) season, but not over the entire sampling period. Summer growth was calculated using differences in TL between the time of last capture and time of marking, divided by the number of days between those corresponding time points. Fall growth was calculated similarly; however, I used the length in October minus the length in August. If a fish was marked during the summer, and caught in August and October it was included in only the fall growth rate calculation, due to the lower sample size of fall recaptures.

All statistical analyses were conducted using SAS (SAS Institute, 1999). All data were checked for normality and transformed if found not to conform to a normal distribution. In the case that data transformation could not be performed, non-parametric analyses were used to examine the data for significant differences. Type I error was controlled when multiple comparisons were performed using Tukey's adjusted p-values. All statistical differences were detected at the $\alpha = 0.05$ level.

Results

Habitat analyses

Channel habitat in the 75 m marking segments showed few differences between urban and rural streams. Urban pools were significantly longer than rural pools (T-test; $t = -1.93$; $df = 23$; $P < 0.05$), but total pool, run, or glide habitat did not differ between urban and rural streams (Table 2, Figure 2). When the frequency of subunit habitats in 75 m was compared between urban and rural streams, no significant differences were found (Table 2). I calculated the ratio of fast to slow water which encompassed a comparison of the extent of riffle to the extent of pools, glides and runs combined. The fast/slow ratio in urban systems was 0.13, which was much lower than the ratio in rural streams (0.33).

Channel dimensions were examined using measurements collected every 15 m throughout the stream reach. Stream width was significantly greater at urban than rural sites (ANOVA; $F = 3.5$; $df = 7, 40$; $P < 0.01$), however differences among the urban sites were also evident. ANAC was significantly wider than CABJ (ANOVA; $t = 2.2$; $df = 40$; $P < 0.05$). Maximum depth of pools used for fish collection was significantly deeper in rural streams (ANOVA; $t = 1.7$; $df = 23$; $P < 0.05$).

Woody debris and rootwads are important sources of habitat for fish in stream channels. The number of instream woody debris was similar, however the number of instream rootwads was significantly greater in rural streams than in urban streams (Table 2, Figure 3). Dewatered rootwads and woody debris were found in higher abundance in both stream categories than instream counterparts. Urban and rural streams had a similar frequency of dewatered rootwads and dewatered woody debris, as well as debris jams along the streambanks (Table 2, Figure 4).

Urban streams exhibited slightly more and higher eroded banks, although the differences between urban and rural streams were not significant (Table 2). Linear length of undercut banks, which provide protective cover for fish, was not significantly higher along rural streambanks as compared to urban streambanks (Table 2). Additionally, the linear extent of bar formation was not significantly different between urban and rural streams (Table 2).

Three out of the four urban streams had a history of some type of stream restoration, which was visually apparent during the stream habitat assessment. Due to these practices, there was a significantly higher linear extent of bank stabilization in urban streams (22 m) than in rural streams that did not display any restoration (ANOVA; $F = 6.1$; $df = 1, 6$; $P < 0.05$). Examples of stabilization include concrete, cobble, and natural fiber netting with native grass plantings. One of the urban streams (LOGU) had 7 m of gabion stabilizing the banks. At the same time, there were many more stormwater and other drain pipes in urban streams than at rural sites (ANOVA; $F = 27.0$; $df = 1, 6$; $P < 0.01$).

Streamflow

Discharge and maximum stage height were measured to compare urban and rural stream habitat associated with fish movement patterns. Baseflow discharge was significantly lower in urban streams when compared to baseflow of rural streams (ANOVA; $F = 9.1$; $df = 1, 34$; $P < 0.01$). The urban baseflow average of June to October monthly measurements was 2.4 x 10-5 ± 5.88 x 10-6 m3/s, but the rural average was 5.5

x 10-5 ± 8.46 x 10-6 m3/s. Additionally, a few sites displayed significant differences. BACK had significantly lower baseflow than HO (ANOVA; t = -4.4; P < 0.001). Over the entire sampling season, rural baseflow discharge was higher than urban baseflow except during the month of September (Figure 5). Maximum stage height was recorded to provide information about the level at which the water level was the highest between sampling days. Data collected in 2004 was not sufficient enough to analyze, due to human interference and the resultant lack of datapoints. The difference between maximum stage height and baseflow stage height (ht-diff) was higher each month in 2005 urban streams than in rural streams, however none of these relationships demonstrated a significant difference (Table 3). Also in 2005, the lowest baseflows were found in June and September during periods of low precipitation, which corresponds to ht-diff calculated for these months (Figure 5, Table 3).

Water quality

Summer water quality was surprisingly similar between urban and rural stream sites, and across years. Dissolved oxygen was higher in rural streams than in urban streams (Table 4; ANOVA; F = 5.71, df = 1, 38; $P < 0.05$). Conversely, there was no significant difference in pH across stream type (Table 4). Stream temperature was not significantly different between urban and rural streams across the sampling season, however conductivity was (Table 4; ANOVA; F = 3.91; df = 1, 38; $P < 0.05$). Interestingly, specific conductivity was higher in rural stream water than expected. Although rural streams exhibited an average conductivity of 0.293 mS/cm compared to 0.399 mS/cm in urban streams, one rural stream site, SENE, demonstrated consistently high specific conductivity measurements that were significantly higher than ANAC (urban; ANOVA; t = -7.8; $P < 0.001$), LIBE (rural; ANOVA; t = -5.4; $P < 0.001$), and MPAX (rural; ANOVA; t = -4.4; $P < 0.001$).

Similarities between urban and rural streams may also be seen throughout the summer on a monthly basis (Figures 6 and 7). Stream temperature was generally higher in urban streams than rural stream regardless of the year sampled, especially during the month of August. October 2004 temperatures were much lower than in 2005 temperatures (Figure 6A). Dissolved oxygen also followed a similar pattern to stream temperature. Rural sites showed higher dissolved oxygen levels than urban sites, even across years (Figure 6B). Specific conductivity was slightly higher in urban than in rural streams, although there was a lot of overlap throughout the sampling season (Figure 7A). Stream sites sampled in 2004 displayed much lower values of specific conductivity in rural streams than in urban streams during the month of August, but not in October. There was no pattern for pH throughout the 2004-2005 sampling seasons. Rural stream pH was higher than urban sites in 2005; however, 2004 urban sites had a higher pH than rural sites (Figure 7B).

Water temperature was recorded by submerged temperature loggers in the 2005 stream sites. Although the logger was lost at MPAX, data from the remaining three temperature loggers indicate that the rural site (SENE) was consistently lower than the temperature regimes at the two urban sites (ANAC and CABJ; Figure 8). Figure 8 illustrates the diel fluctuations in stream temperature, corresponding to fluctuations in air temperature. There is also a noticeable seasonal pattern indicating a slow decline in water temperature beginning at the end of August, and after a few peaks, continuing throughout September and October (Figure 8). During the week of August 8^{th} to August

14th, the stream temperature in CABJ was much higher than either the ANAC or SENE. This was also noted during the August fish recapture, when the stream temperature was recorded at 26.50°C. During this time, stream temperature ranged from a minimum of 23.95°C to a maximum of 28.10°C.

Species richness

Species collected at rural sites in addition to BND and CKB included: Blue Ridge sculpin *Cottus caeruleomentum*, white sucker *Catostomus commersoni*, rosyside dace *Clinostomus funduloides*, longnose dace *Rhinichthys cataractae*, cutlips minnow *Exoglossum maxillingua*, common shiner *Luxilus cornutus*, American eel *Anguilla rostrata*, tessellated darter *Etheostoma olmstedi*, shield darter *Percina peltata*, bluegill sunfish *Lepomis macrochirus*, green sunfish *Lepomis cyanellus*, redbreast sunfish *Lepomis auritus*, pumpkinseed sunfish *Lepomis gibbosus*, and yellow bullhead *Ameiurus natalis*. Fish species collected at urban sites included (in addition to BND and CKB): white sucker *C. commersoni*, rosyside dace *C. funduloides*, longnose dace *R. cataractae*, central stoneroller *Campostoma anomalum*, American eel *A. rostrata*, bluegill sunfish *L. macrochirus*, green sunfish *L. cyanellus*, pumpkinseed sunfish *L. gibbosus*, largemouth bass *Micropterus salmoides*, and goldfish *Carassius auratus*. Species richness at urban and rural sites was not significantly different (5.4 ± 0.6 vs. 7.4 ± 1.4; T-test; $t = -1.35$, $df = 4$, $P = 0.12$).

Mark and recapture

In 2004, I marked 341 fish (220 BND and 121 CKB), and 750 fish (566 BND and 184 CKB) in 2005. All fish that were collected in the initial sampling period in 2004 were marked, however additional fish were marked during subsequent recaptures in 2005. I marked slightly more fish in urban streams (BND = 471, CKB = 192) than in rural streams (BND = 315, CKB = 113), although there was no significant difference in the number of fish marked per site across years (ANOVA; F =1.67; df = 1, 6; P = 0.24). Similarly, there was no statistical difference in the number of BND or CKB marked in urban and rural streams. However, due to my marking program, I marked significantly more fish per stream in 2005 than in 2004 (188 vs. 85; ANOVA; F = 11.61; df = 1, 6; P < 0.05).

Recapture rates varied with both species and stream category. Creek chub had a higher recapture rate than BND and I recaptured more fish in rural streams than in urban streams (Table 5). In urban streams, 16% of all BND and 31% of CKB were recaptured while rural streams had a higher recapture rate by species (BND = 22%, CKB = 44%; Table 5). Three percent of recaptured fish were captured more than once (urban: 20 fish – recaptured 2x, 1 fish – recaptured 3x; and rural: 10 fish – 2x, 1 fish – 3x).

There was no difference in the number of recaptured fish between urban and rural streams (ANOVA; F = 0.08; df = 1, 6; P = 0.78), nor between sites (ANOVA; F = 0.62; df = 6, 1; P = 0.75). I recaptured more BND in 2005 than in 2004 (25 vs. 8; ANOVA; F = 18.06; df = 1, 6; P < 0.01), but there was no statistical difference in CKB among years. Additionally, the number of recaptures of each fish species was not found to differ between sites (ANOVA; BND: F = 1.04; df = 1, 6; P = 0.93; CKB: F = 0.01; df = 1, 6; P = 0.93). Finally, during 2005 when I sampled an extra 50 m on each end of the 275 m recapture segment of CABJ and MPAX, I found no marked fish.

Abundance

A Jolly-Seber open population model was used to estimate population size within each stream site. When estimates from within each year and site were averaged, there was no difference in the number of BND in urban and rural streams (ANOVA; F= 4.4; df = 1, 6; $P = 0.08$; Figure 9). This pattern was also seen in urban and rural CKB (ANOVA; F = 3.9; df = 1, 6; $P = 0.10$). However the overall abundance of BND was significantly higher than that of CKB (ANOVA; F = 6.5; df = 1, 14; $P < 0.05$; Figure 9). When these species were combined, I found no difference between urban and rural populations (ANOVA; F = 4.3; df = 1, 14; $P = 0.06$).

Movement patterns

Streams in both urban and rural categories were comprised of both movers and stayers. Fish in urban streams were more likely to move out of their home pools, while rural fish were more likely to remain in their home pool throughout the sampling period (Figure 10). Fifty-eight percent of urban fish were classified as movers, while 42% were stayers (Figure 10). Conversely, rural streams exhibited a greater (60%) proportion of residents than urban streams, while the remaining 40% were found to move to other stream pools (Figure 10). The proportions of movers and stayers did not differ between rural and urban streams (G-test; G = 2.8; df = 1; $P = 0.09$), though the difference was marginally non-significant.

Urban BND and CKB also moved farther than rural conspecifics. The distance traveled by urban mover subpopulation ranged from 10.5 m to 133.5 m outside of their home pool area, while the rural mover range was from 9 m to 97.5 m (Figure 11). Movement by one urban fish was estimated at 157.5 m, which may be an outlier compared to the rest of the movers (Figures 11 and 12A). Urban fish moved significantly farther within the stream reach than did rural fish (Table 6; Kruskal-Wallis; χ^2 = 13.17; P < 0.001;). Rural fish displayed a median distance moved equal to 0 m compared to 13 m for urban fish (Table 6).

Individual sites were evaluated to determine if movement patterns were similar in streams grouped by percent land use in the upstream watershed. Rural sites showed significantly different distance moved (Kruskal-Wallis; χ^2 = 8.02, P < 0.05). Fish at HO moved the farthest (34.5 m) from their home pool, while SENE fish moved the least (8.8 m). Conversely, fish movement in urban streams did not differ significantly across sites (Kruskal-Wallis; χ^2 = 0.77; P = 0.86; range = 19.6 to 34.0 m). Blacknose dace moved greater distances in urban streams than in rural streams (37.7 vs. 13.0 m; Kruskal-Wallis; χ^2 = 13.17; P < 0.001). On the contrary, movement was not significantly different between stream types for creek chub (urban = 25.0 m, rural = 14.7 m; Kruskal-Wallis; χ^2 =2.07; P = 0.15).

There was no difference in monthly movement (July, August, October) in either rural streams (Kruskal-Wallis; χ^2 = 0.072, df = 2, P = 0.96) or in urban streams (Chi-square; χ^2 = 0.087, df = 2, P = 0.96). In both 2004 and 2005, urban fish moved greater distances than fish in rural streams. Distance traveled by fishes was significantly lower in rural (19.8 m) than urban in 2004 (28.2 m; Kruskal-Wallis; χ^2 = 5.1, P < 0.05) and in 2005 (rural = 9.3 m, urban = 33.9 m; Kruskal-Wallis; χ^2 = 6.6, P < 0.01).

Fish in urban streams demonstrated greater variability in movement away from their home pool, while distances moved by rural fish were clumped (Figure 12). Clumps appear between 10 and 30 m, 45 and 60 m, 75 m, and 90 and 100 m (Figure 12B).

Diffusion, as measured by the standard deviation of the signed movement distribution (Petty and Grossman 2004), in urban stream fish was higher than in rural stream populations (Table 6). The distribution of movement distances in rural populations was more leptokurtic than urban populations (Table 6). Movement by urban cyprinids appeared to be more or less equal up and downstream, whereas rural fish had a tendency to move farther upstream than downstream (Figure 12).

Home range was estimated for each fish that had been marked and recaptured at least once, thus including movers and stayers (rural = 106, urban = 115; Table 7). Estimation of the home range was limited by pool size, but could have a maximum range of the entire 275 m sampled. Mean home range of urban fish was significantly larger than that of rural fish (ANOVA; F = 35.3; df = 1, 217; $P < 0.001$). The maximum home range of urban fish was also slightly larger than rural fish (urban BND, CKB = 173, 155 m vs. rural BND, CKB = 152, 104 m; Table 7). When home range was analyzed by species, BND and CKB home ranges were similar across stream category (ANOVA; F = 2.3, df = 1, 217; $P = 0.13$; Table 7). However, within each species, significant differences were observed between urban and rural populations. Urban BND had significantly larger home ranges than rural BND (Table 7).

Length and growth analyses

In rural streams, there were no significant differences in TL of mover and stayer BND (ANOVA; F = 3.1; df = 1, 62; $P = 0.08$) or CKB (ANOVA; F = 0.35; df = 1, 40; $P = 0.60$). Rural BND ranged in size from 45 – 80 mm, while CKB ranged in size from 53 – 230 mm. Urban mover BND were significantly longer than stayers (ANOVA; F = 9.9; df = 1, 45; $P < 0.01$; Figure 15). The average length of urban mover BND was 60.2 mm with a range of 50 – 71 mm (n = 44), while stayers ranged between 52 – 65 mm. Urban creek chub, however, did not show a significant difference between movers and stayers (ANOVA; F = 0.54; df = 1, 66; $P = 0.47$). Urban CKB movers ranged in length between 51 and 184 mm, with an average of 104.3 mm (n = 23). The range of urban CKB stayers was 57 – 150 mm.

Growth rates calculated for all fish that were marked and recaptured at least once displayed species differences. Growth rates were higher in CKB than in BND (ANOVA; t = -4.6; df = 186; $P < 0.001$; Table 8). There was no difference between urban and rural fish growth rates when species were combined, however rural CKB grew significantly more than urban CKB (ANOVA; t = 2.22; df = 186; $P < 0.05$; Table 8). Growth analysis of each species by season did not indicate any significant differences (Table 8). However, CKB growth rates were always slightly higher in the summer (Table 8).

There were no significant relationships between total length (TL) and distance moved when data included both movers and stayers. Using total length data from only mover BND and CKB, length did not predict distance moved over the sampling period for rural length. Neither rural BND (Linear regression; Adj.-$R^2 = -0.04$; $P = 0.73$) nor CKB (Linear regression; Adj.-$R^2 = 0.02$; $P = 0.27$) movement was related to total length measured at the time they were marked. However, in urban streams, both species displayed significant relationships between movement and TL. BND exhibited a slightly negative (slope = -0.023) relationship between $\log_{10}$ movement and TL (Linear regression; Adj.-$R^2 = 0.09$; $P < 0.05$; Figure 13). Alternatively, length positively (slope = 0.004) predicted distance moved in urban creek chub (Linear regression; Adj.-$R^2 = 0.19$; $P < 0.05$; Figure 14).

Growth in movers and stayers was not different in BND (ANOVA; F = 0.77; df = 1, 113; P = 0.38) or CKB (ANOVA; F = -0.01; df = 1, 77; P = 0.94). Since there was no difference in growth by season in BND and CKB (Table 8), seasonal data were pooled to examine the relationship of growth on distance moved by species and stream category. Rural BND displayed negative daily growth the more they moved (Linear regression; Adj.-R^2 =0.24; df = 1, 24; P < 0.01; Figure 16), however no meaningful relationships were found for urban BND (Linear regression; Adj.-R^2 = 0.22; df = 1, 12; P = 0.51), rural CKB (Linear regression; Adj.-R^2 = -0.03; df = 1, 32; P = 0.93), or urban CKB (Linear regression; Adj.-R^2 = 0.14; df = 1, 17; P = 0.06). The growth rate of one individual that was particularly high (0.41 mm/d) during the summer season was found at MPAX (2005; Figure 16). This fish was marked in June and recaptured once in August, growing 23 mm during this time period.

Finally, to determine if stormflow or maximum stage height was a mechanism behind movement by BND and/or CKB, monthly distance moved was regressed upon the difference in stage height (ht-diff). Although many of these relationships were not significant, there was a positive relationship between CKB movement and ht-diff in urban streams during the 2005 sampling season (Table 9).

Discussion

This study of two stream cyprinid species attempted to distinguish differences in movement patterns across populations found in streams draining urban and rural watersheds. I hypothesized that the proportion of movers and stayers would differ between rural and urban fish populations. Secondly, I hypothesized that fish in urban streams will demonstrate a larger home range than rural fish in pursuit of suitable habitat.

Results presented here show significant differences in urban and rural fish movement patterns. Urban fish are more likely to move about the stream reach, exploring new territory in search of habitat, food resources, or potentially protection during high flows. These data also provide evidence that both movers and stayers exist within fish populations, but that urban streams have a greater proportion of movers than stayers. In addition, urban fish move farther distances upstream and downstream from where they were originally captured. Conversely, rural fish are more likely to be found in the same pool on repeated occasions. Rural fish populations are composed of more stayers than movers and the distance moved by rural movers was also less than urban movers.

Estimation of home range has been applied to a number of fish movement studies (Goforth and Foltz 1998, Hill and Grossman 1987, Petty and Grossman 2004, Colyer et al. 2005). In fish populations, a home range may include the area in which individuals forage, hide from predators, mate, build and protect nests, and rest. In the case of pool-dwelling stream cyprinids, a home range is composed of one or more connected pools within a stream reach. Other studies on movement of stream cyprinids indicate similar home ranges to those estimated in this study. Goforth and Foltz (1998) estimated the home range of the yellowfin shiner *Notropis lutipinnis* as 42.9 ± 79.0 m, while Hill and Grossman (1987a) reported a home range for the rosyside dace as 19.3 ± 8.0 m. Some species, such as the bluefin shiner *Cyprinella caerulea* moved an average distance of 137.0 m (Johnston 2000), which is much higher than BND and CKB in either of my rural or urban streams.

In this study, home range estimates support differences within rural and urban fish populations. Rural fish used a smaller complex of habitat patches than urban fish, and some may have only used one habitat patch. Because pools are separated many times by riffles and/or runs, I considered each pool equal to a patch. Therefore, average patch area (based on length and width estimates) was smaller in rural streams, and it is likely that food resources are more plentiful here as well. Rural stream baseflow was also higher, presumably delivering more high quality drifting insects and other macroinvertebrates to pool habitats. Although habitat patches were smaller, fish may not need to move out of a pool to forage if uptake by individuals does not exceed input into the patch.

Urban stream ecosystems display the opposite circumstances. Patch area was relatively large, however, benthic food resources were poor and the lower baseflow may not have delivered sufficient food items. BND and CKB are generalists and will forage on almost anything from insects, worms, and arachnids, to larval fish, algae, and detritus (Jenkins and Burkhead 1993). Even though these fish will eat a wide variety of food items, urban fish will move out of a habitat patch when resources are depleted to forage for better food quality and quantity. Optimal foraging theory or the marginal value theorem asserts that individuals will move out of a habitat when food resources are equal to or less than the quality of resources in surrounding patches (Charnov 1976). Although rural fish may encounter similar scenarios where food resource quality equals or is less than the quality of adjacent habitat patches, abundance of fish was higher in urban pools. This means that urban patches of initially low food quality were under foraging pressure of potentially twice as many individuals based on the population analysis as compared to patches in rural streams composed of higher food quality. Ideal free distributions have been used to describe the movement of stream fish between habitat patches as well as the overall dispersal of individuals with respect to resource availability (Giannico and Healey

1999). This model predicts equal rates of return for all fish individuals when patches differ in resource availability. Therefore, urban fish may have expanded their home range to include more pools/patches in order to sufficiently match their ecological niche requirements.

The shape of a movement distribution provides information on the overall pattern of dispersal. The rural movement distribution of BND and CKB was clumped, whereas urban fish population movement may be better described as diffusive. Petty and Grossman (2004) explained fish movement as the average of unsigned distance traveled, and diffusive spread as the standard deviation of signed movement. Variability in movement is become a good statistical method to assess patterns of mobility, especially when studying differential environmental conditions (Rodríguez 2002, Petty and Grossman 2004, Skalski and Gilliam 2000). Urban fish move on average almost three times farther as rural fish and diffuse throughout their habitat twice as much as rural fish. The shape or kurtosis of the movement distributions demonstrate higher affinity or pool residence in rural streams relative to urban streams, especially when comparing similar sample sizes of fish in each stream category. Previous studies on small stream fish populations have indicated that movement patterns of bluehead chubs are leptokurtotic, whereas creek chub follow a more normal distribution over summer months (Skalski and Gilliam 2000). In the present study, creek chub distance data was not normal in urban or rural streams; however, movements in a rural stream displayed a more leptokurtic distribution than urban fish, due to the high number of fish who did not move. In addition, rural fish demonstrated similar upstream bias to other studies (Skalski and Gilliam 2000), whereas urban fish disperse equally up and downstream. During summertime baseflow, rural fish may be able to not only sense increasing flow but move through the channel to upstream refugia without getting washed downstream. Urban fish may not be able to withstand the forces of turbulent stormflow, heading downstream. Although I was not able to estimate maximum stormflow discharge between sampling periods, the height difference between maximum stage height and baseflow stage height at urban streams was much greater than in rural streams. This evidence that urban stream fish experience drastically different physical conditions than fish in rural streams may be enough to suggest that flow regime may be responsible for diffusion of urban stream fish.

Although many urban streams may look similar to one another, they are not all alike. The estimate for BND population abundance in CABJ was significantly lower than estimates for other urban streams. CKB estimates for CABJ were similar to other urban and rural streams; however, if CABJ had more BND, the analysis would have indicated that urban streams have two to three times more biomass than rural streams. Recent studies have demonstrated that small first order urban streams in the eastern Piedmont are dominated by BND, and that abundance of BND can be greater than 200 individuals per 75 m in highly urbanized streams (>75% urban land use; Morgan and Cushman 2005). One reason for this dominance among the fish assemblage is that urban streams have significantly lower species richness as ULU increases in the watershed (Morgan and Cushman 2005, Paul and Meyer 2001, Weaver and Garman 1994). This is most likely due to the elimination of pollution intolerant species, allowing tolerant generalist species like BND and CKB to capitalize on available habitat and resources (Walters et al. 2003, Scott 2006).

Although significant relationships did not occur in rural stream populations between length at marking and movement throughout the summer, the analyses indicated that length of urban fish does relate to their movement behaviors. The lack of a relationship in rural CKB is similar to findings by Smithson and Johnston (1999). However, longer CKB moved farther distances within the urban streamscape. Urban BND displayed a very different relationship between length and vagility. When BND were divided into movers and stayers, the data suggested that stayers were smaller in total length than movers. Results from linear regression of only the mover subpopulation implied a negative relationship between length at time of marking and distance moved. Although this was a very weak relationship, the smaller the individual, the farther it traveled throughout the summer. Therefore, when the two subpopulations were combined, a bell-shaped curve was generated. Stayers, who do not move, are the smallest individuals in the population creating the left tail. Within the mover subpopulation, the larger individuals move the least (peak), but the smaller the fish, the more it moved (right tail). This model creates a favorable body size for greatest movement potential throughout the stream reach around 60 mm. Thus, fish found in the tails of the movement distribution are not as long as the fish that are found in the peak.

Mechanisms behind movement

As a result of this fish movement study, I propose three different mechanisms behind the observed pattern of movement in urban streams involving environmental and/or ecological roles. 1) Urban stream populations may be responding to the harsh flow regime that has become so common in urbanized watersheds, and movement reflects displacement from high stormflows. 2) Habitat differences between rural and urban channels were also evident, and may provide a direct relationship between habitat degradation and diffusion within urban fish populations. 3) Ecological interactions such as intraspecific and interspecific competition could be responsible for differences in fish abundance at urban and rural streams, and therefore cause greater movement in urban populations.

One reason for a high degree of dispersal among urban fish populations may be related to the altered flow regime found in urban stream networks. Roy et al. (2005b) found that hydrologic variables explained up to 66% of the variation in the composition of and abundance of urban fish assemblages. Urban streams have lower baseflow than rural streams; however, during storms, peak stormflow reaches greater maximum stage height in channels that are wider than those in rural environments. In many urban watersheds, reduced baseflow is due to a disconnection between the groundwater and the channel, contributing to greater fluctuations in diel temperature and dissolved oxygen levels (Brasher 2003, Groffman et al. 2003, Walsh et al. 2005a). Thus, urban BND and CKB may have adapted to extremely dynamic flow conditions. Fish that move downstream during a storm may move back upstream after the flow subsides. However, summertime baseflow in urban streams may be low enough to create barriers to upstream movement after stormflow moves the fish downstream. Other studies have established that riffles can become barriers to movement in small streams during summertime baseflow (Lonzarich et al. 2000). It is not uncommon for pools to become isolated during summer drought, especially in urban streams (personal observation). Therefore, movement up or downstream may become more permanent in urban systems until the next storm increases the water level in the stream, as compared to rural systems where movement can still occur between pools during baseflow.

The availability of fish habitat within a stream reach is a significant aspect of habitat use and movement to other habitat patches. Although my study presented few significant relationships between habitat and stream category, there are a variety of features of fish habitat that were not measured. Basic water quality parameters were measured, however other parameters such as nutrients, heavy metals, oils, bacterial loads, and other pollutants may provide important information about the environmental quality of baseflow as well as stormflow. Water temperature is an important quality that was monitored at two urban and one rural stream throughout summer 2005, and gave insight to daily patterns in these streams. The sustained peak in temperature in CABJ was surprising but may have given fish reason to move to deeper pools or other reaches where the temperature was cooler. Other instream habitat characteristics such as the frequency of instream rootwads and other instream structures provide stable refugia before, during and after storms. Rural fish movement may have resembled a restricted movement pattern due to the higher number of refugia as compared to urban streams and their fish movements.

Finally, urban stream populations may be experiencing a high degree of competition. In rural streams, where pool size is smaller and fish abundance is low, intraspecific competition for food and space may be minimal. The urban streams that were selected for this study had a low average BIBI (2.3 versus 3.5 for rural), which is the difference between poor and good benthic health (based on the MDDNR IBI scale).

Although BND and CKB are generalists, this low quality forage resource may provide competitive pressure among individuals within each pool.

More fish were marked in urban streams because more BND and CKB were collected in the 75 m marking segment than in rural streams. Many studies have found that species richness declines as the watershed becomes more urban (this study, Morgan and Cushman 2005, Paul and Meyer 2001, Weaver and Garman 1994). Rural streams may have higher species richness, however those assemblages are likely not competing for the same resources compared to a fish community dominated by BND and CKB. In a simplified fish assemblage of stream cyprinids, intraspecific competition may play a bigger role than interspecific competition. Only one species (longnose dace) that prefers riffle and run habitat was collected in the urban streams in this study as compared to rural streams which presented four species (longnose dace, Blue Ridge sculpin, tessellated darter, and shield darter). In a study of stream salmonids, fish selected and moved to habitats outside of their home pool when optimal foraging positions were taken (Gowan and Fausch 2002). Since urban fish used a larger home range and moved out of their home pool more frequently than rural fish, this may be evidence that these feeding generalist species use foraging positions. Nakano (1995) showed that dominant salmonid individuals were more sedentary than their subordinates and occupied the best feeding positions in a pool. Since there is considerable spatial niche overlap of BND and CKB, competitive interactions may play an important role in movement of urban assemblages. Therefore, when habitat patch quality is low and abundance is high, urban fish move to other stream pools in an attempt to alleviate competition and decrease time spent foraging.

Conclusions

Little is known about the movement patterns of stream cyprinids, let alone differences among and between cyprinid populations. Although cyprinids are not considered game species, they are an integral part of the stream food web and therefore create links to predatory game species that are of interest to anglers and fisheries managers. This research on fish movement in contrasting stream environments has provided insight to mechanisms behind differences between mover and stayer populations, fish ecology in degraded systems, and essential information about their life history strategies. Similar to other studies of stream fish, populations are divided into those individuals that display restricted movement and those who actively explore and use a larger stream reach (Hilderbrand and Kershner 2000, Petty and Grossman 2005, Skalski and Gilliam 2000, Colyer et al. 2005, Gowan et al. 1994). This proportion may vary depending on ecological and environmental conditions within the stream ecosystem. This study suggests that urban fish populations have a greater proportion of movers than stayers, and utilize a larger home range, dispersing throughout long stream reaches over the course of a few months, than rural fish populations. Species-specific patterns were also evident in urban streams. Larger mover CKB swam greater distances than smaller mover CKB. BND movers were longer than stayers, suggesting that ecological niche requirements were fulfilled using a widespread home range. Thus, differential movement patterns do exist in urban and rural streams potentially due to environmental and or ecological mechanisms.

Tables

Table 1. First order stream sites used in this movement study (2004-2005) were previously sampled and characterized in the Maryland Biological Stream Survey (MBSS). Site names are from original MBSS sites. Basin abbreviations relate the major river basin each site is found within. Percent urban land use (%ULU) and watershed area were calculated from digitized maps. Latitude and longitude are presented in decimal degrees. The number of blacknose dace (BND) and creek chub (CKB) presented here are the total number of fish marked at each site. *Ucat = Urban Category; PATX = Patuxent, PATP = Patapsco, LGUN = Lower Gunpowder, BACK = Back, POTM = Potomac.*

Site Name	Basin	UCat	Name	%ULU	Watershed Area (ha)	Latitude	Longitude	Year	BND	CKB
HO	PATX	Rural	Benson Branch	1.21	421	36.2640	-76.9550	2004	53	0
LIBE	PATP	Rural	Keysers Run	5.64	161	39.4697	-76.8593	2004	30	57
LOGU	LGUN	Urban	Jennifer Branch	64.21	267	39.4040	-76.5110	2004	62	19
BACK	BACK	Urban	Stemmers Run	64.86	347	39.3670	-76.5229	2004	75	45
SENE	POTM	Rural	Magruders Branch	13.11	277	39.2900	-77.2120	2005	115	18
MPAX	PATX	Rural	Unamed Trib	0.00	129	39.1945	-76.9610	2005	117	38
CABJ	POTM	Urban	Cabin John Creek	72.03	238	39.0714	-77.1518	2005	96	116
ANAC	POTM	Urban	Sligo Creek	62.55	367	39.0226	-77.0307	2005	238	12

Table 2. Stream habitat in urban and rural streams divided into channel units, fish habitat, and sediment characteristics. Significant differences between urban and rural streams are indicated by $P < 0.05$.

Channel Units	Urban	Rural	F	Df	*P*
Total length					
Riffle	8.8	20.3	5.4	1, 6	0.06
Pool	25.3	35.3	2.6	1, 6	0.16
Run	29.3	22	0.86	1, 6	0.39
Glide	12	3.3	0.91	1, 6	0.38
Number					
Riffle	3	5	4.2	1, 6	0.09
Pool	4	4	0.86	1, 6	0.39
Run	3	3	0.07	1, 6	0.80
Glide	2	1	0.50	1, 6	0.51
Fish Habitat					
Instream WD	3	4	0.22	1, 6	0.66
Instream RW	2	4	12.8	1, 6	< 0.05
Dewatered WD	6	7	0.12	1, 6	0.74
Dewatered RW	8	7	0.05	1, 6	0.83
Debris Jams	5	7	0.43	1, 6	0.54
Sediment					
Linear erosion	110	74.4	4.5	1, 6	0.08
Max ht. of erosion	1.7	1.1	3.7	1, 6	0.10
Undercut banks	5.5	23	3.9	1, 6	0.10
Bar formation	63.3	55.5	0.49	1, 6	0.51

Table 3. Difference between maximum stage height and baseflow stage height in 2005 urban and rural streams. Stage height was collected using a piezometer placed in the stream substrate, secured by woody debris or a rootwad, and was measured on a monthly basis from June to October. Baseflow height was indicated by the water level at the time of sampling, and maximum height was recorded as the highest water level that was reached since the previous sampling account. The difference (Ht-diff; m) between these levels was examined to detect differences between 2005 rural (R) and urban (U) streams by month.

Month	Ht-diff	F	df	*P*
June	R- 0.065 U- 0.175	0.50	1, 2	0.55
July	R- 0.285 U- 0.645	2.44	1, 2	0.26
August	R- 0.415 U- 0.925	8.17	1, 2	0.10
September	R- 0.055 U- 0.135	0.35	1, 2	0.62
October	R- 0.530 U- 0.660	0.22	1, 2	0.69

Table 4. Water quality at urban and rural stream sites, presented as the average over all sites. Dissolved oxygen (mg/L), pH, temperature (°C), and specific conductivity (mS/cm) were measured each time a site was visited. Significant differences are shown with a $P < 0.05$.

Parameter	Urban	Rural	F	df	*P*
Dissolved Oxygen	6.51	7.62	5.71	1, 38	< 0.05
pH	7.00	7.02	0.05	1, 38	0.83
Temperature	19.92	18.34	2.24	1, 38	0.14
Specific Conductivity	0.399	0.293	3.91	1, 38	< 0.05

Table 5. Recapture efficiency presented as the rate of recapture in urban and rural streams, and by species. The numbers outside of the box represent the overall recapture rate by stream category (horizontally) and species (vertically). The numbers inside the box represent the recapture rate for each stream category, broken down by species. All rates indicate the percentage of fish that were recaptured at least once.

		Rural	**Urban**
	Total	28	21
BND	36	44	31
CKB	18	22	16

Table 6. Signed and unsigned movement parameters in urban and rural streams. The mean, standard deviation, and kurtosis are estimates of signed distance, while the mean, standard error of the mean (SEM), and median are estimates of unsigned distance. Signed SD of distance is a measure of diffusion. Mean unsigned distance is a measure of movement.

	Signed Distance				Unsigned Distance	
	N	Mean	Std Dev	Kurtosis	Mean (± SEM)	Median
Urban	115	6.2	51.21	1.05	32.1 (3.74)	13
Rural	106	7.7	26.65	3.61	13.7 (2.34)	0

Table 7. Estimates of home range for urban and rural blacknose dace (BND) and creek chub (CKB). Home range was calculated using the distance between the most upstream and most downstream capture pools plus the length of each of those pools.

Species	Type	N	Mean (m)	SEM (m)	t-value	df	*P*
BND	Rural	64	26.6	3.98	-4.9	217	<0.001
	Urban	68	51.5	4.87			
CKB	Rural	42	24.1	4.35	-3.7	217	<0.01
	Urban	47	42.9	6.15			
Combined	Rural				0.86	217	0.83
Combined	Urban				0.20	217	0.57

Table 8. Growth rates (GR) of blacknose dace (BND) and creek chub (CKB) calculated by stream type and season. Summer GR is the growth incurred between June and August, while Fall GR is growth incurred between August and October. Growth rates were calculated by taking the difference in total length (mm), and dividing by the number of days between capture and recapture. GR (mm/d) was calculated for all fish that were recaptured at least once.

	Type	Season	N	GR Mean	GR SEM	t-value	*P*
CKB	Rural		41	0.25	0.040	2.22	< 0.05
	Urban		38	0.19	0.032		
BND	Rural		61	0.03	0.013	-0.40	0.69
	Urban		54	0.06	0.012		
CKB	Rural	Summer	24	0.26	0.046	-0.38	0.70
		Fall	17	0.24	0.075		
	Urban	Summer	33	0.20	0.036	-1.65	0.10
		Fall	5	0.07	0.038		
BND	Rural	Summer	54	0.03	0.015	-0.01	0.99
		Fall	7	0.03	0.007		
	Urban	Summer	45	0.06	0.015	-0.17	0.87
		Fall	9	0.05	0.009		

Table 9. Movement of blacknose dace (BND) and creek chub (CKB) regressed on stage height difference (ht-diff) in 2005 urban and rural streams. Monthly movement estimates of each species was related to the difference between baseflow and maximum stage height for July, August and October to determine if movement was correlated or a cause of stormflow between recapture visits.

BND	F-value	Df	P-value	Adj-R^2
Rural	0.10	1, 4	0.37	-0.0002
Urban	0.34	1, 4	0.34	-0.15
CKB				
Rural	0.14	1, 4	0.73	-0.27
Urban	7.22	1, 4	0.07	0.61

Figures

Figure 1. Map of the stream networks in Maryland. Sampling sites for 2004-5 are denoted by red (urban) and yellow (rural) dots on the map.

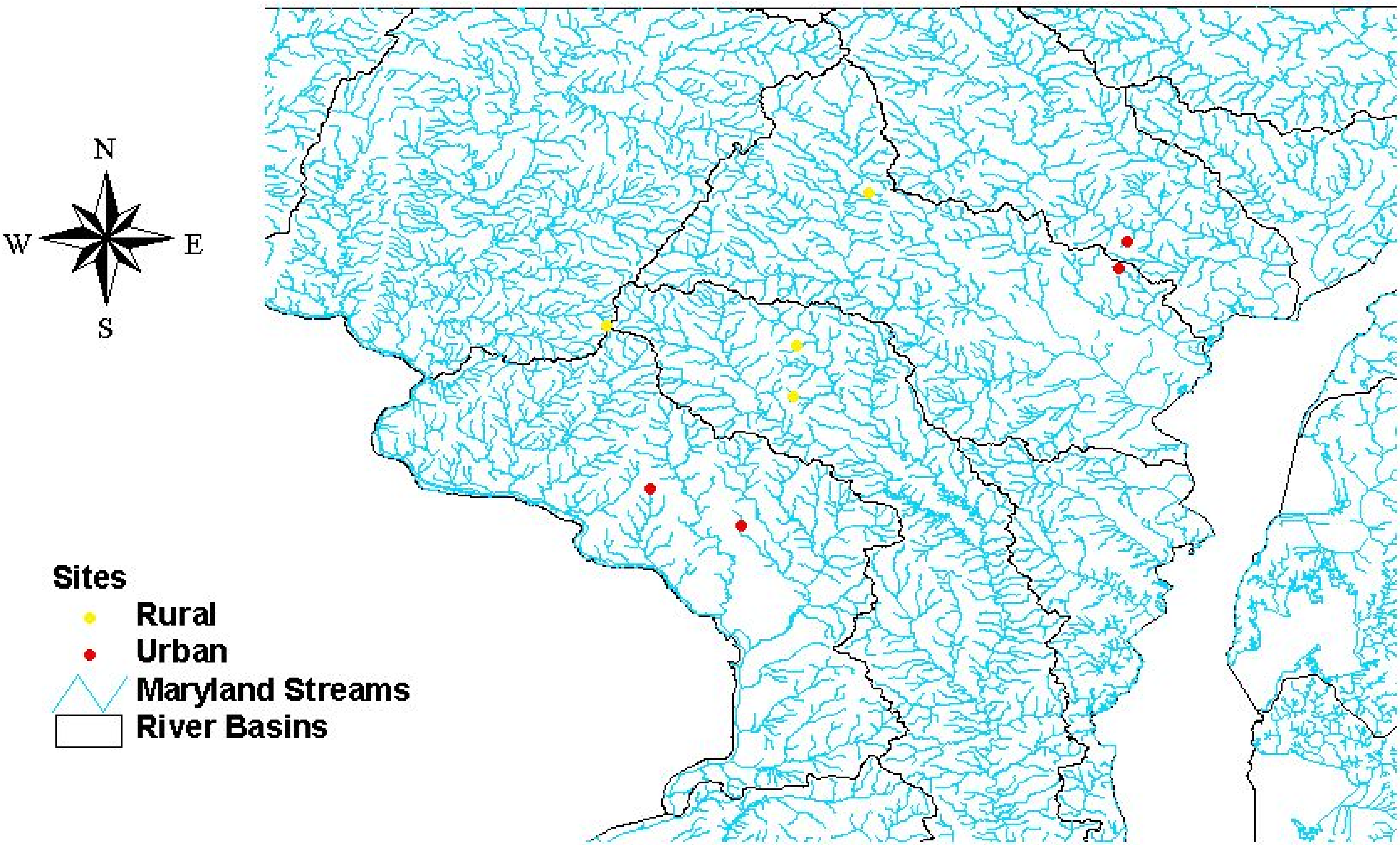

N
W
E
S
Sites
Rural
Urban
Maryland Streams
River Basins

Figure 2. Stream channel habitat subunit composition in each urban and rural stream sampled. Linear extent of habitat subunits was measured within the 75 m habitat and fish marking segment, and converted to percentages. It was possible to have more than 75 linear m of all subunits combined if the channel was split by a center bar or if the stream was wide enough to form two types of habitat. The top four sites were urban streams, while the lower four sites were rural. The site name indicates the year sampled.

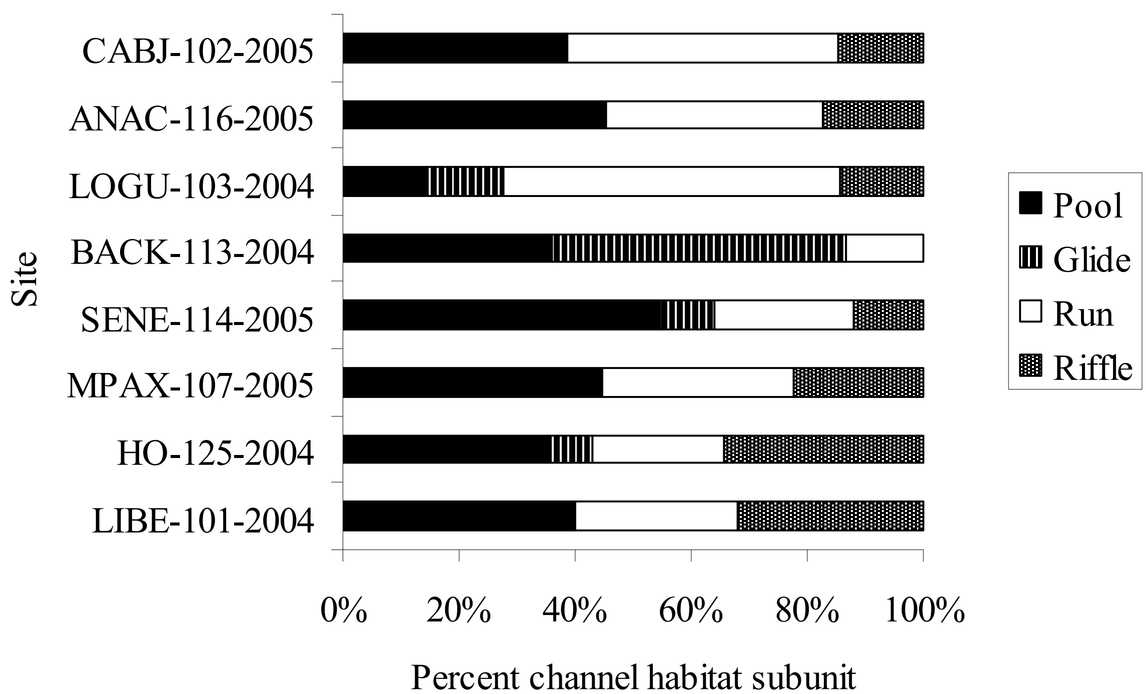


Figure 3. Comparison of instream rootwads and woody debris in urban and rural stream habitat. Instream structures offer protection and food sources for fish during baseflow. Rural streams displayed significantly more instream rootwads than urban streams. * indicates a significant difference at $p < 0.05$, and bars represent mean ± SEM.

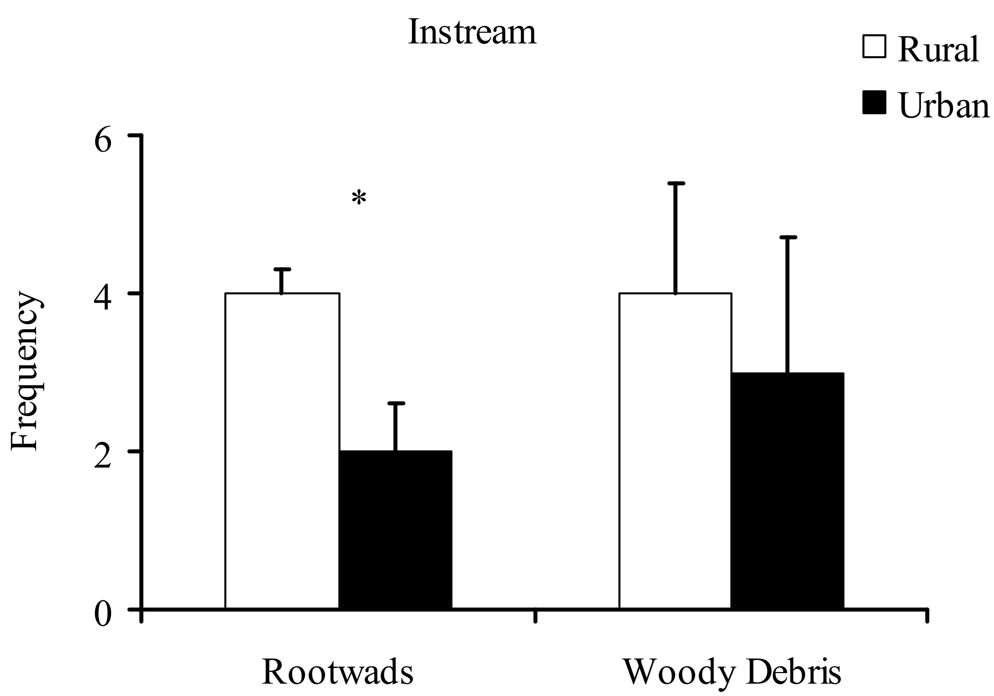


Figure 4. Comparison of dewatered rootwads and woody debris in urban and rural stream habitat. Dewatered habitat structures have the potential to provide protection and refuge during high stormflows. There was no significant difference ($P > 0.05$) between urban and rural streams for either rootwads or woody debris. Bars represent mean ± SEM.

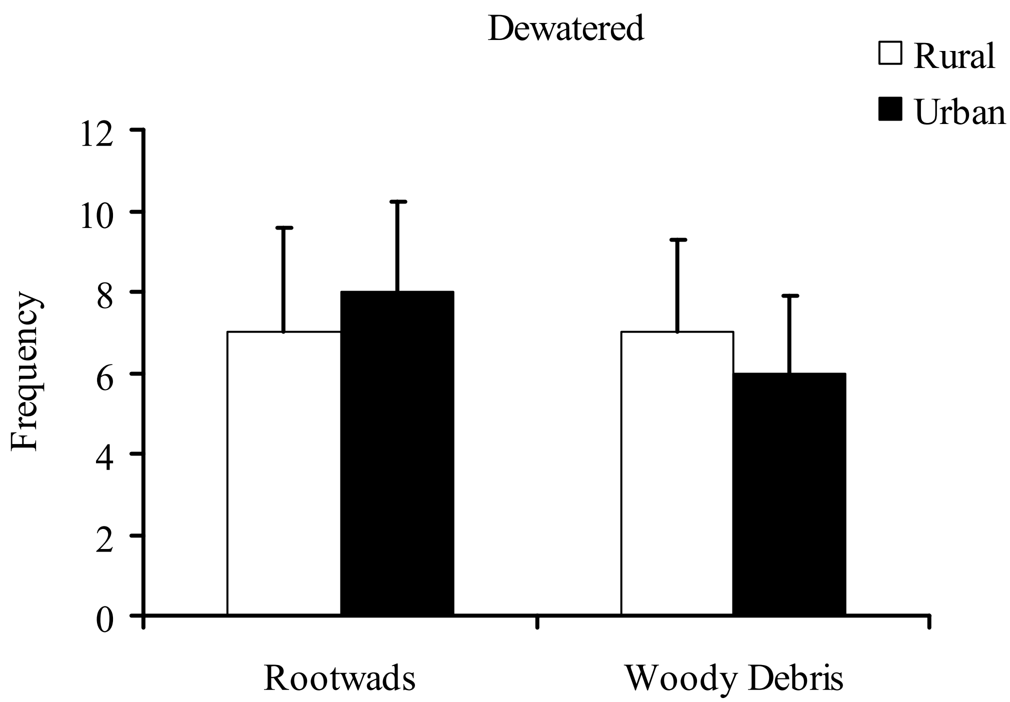


Figure 5. Baseflow discharge measured throughout the fish sampling season was higher in rural streams than in urban streams. Discharge was measured on an approximately monthly basis, in the same location within each stream. Each data point represents the average of four streams in July, August, and October (2004 and 2005 sites), but an average of two streams each for early June, late June, and September (2005 sites). Bars indicate standard error about the mean. Baseflow discharge (m^3/s) was normalized for watershed area (ha).

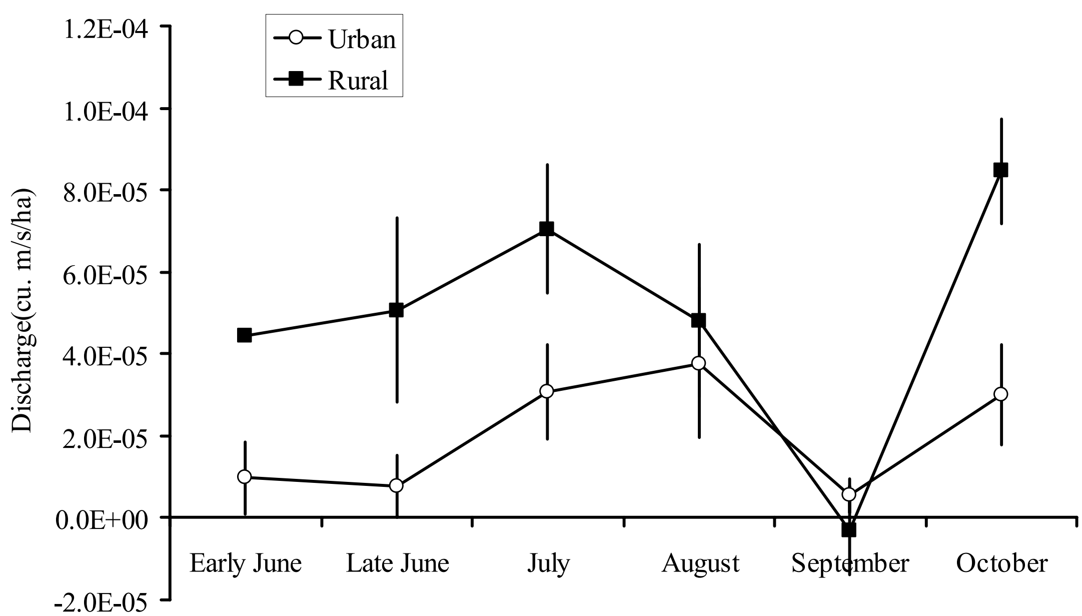
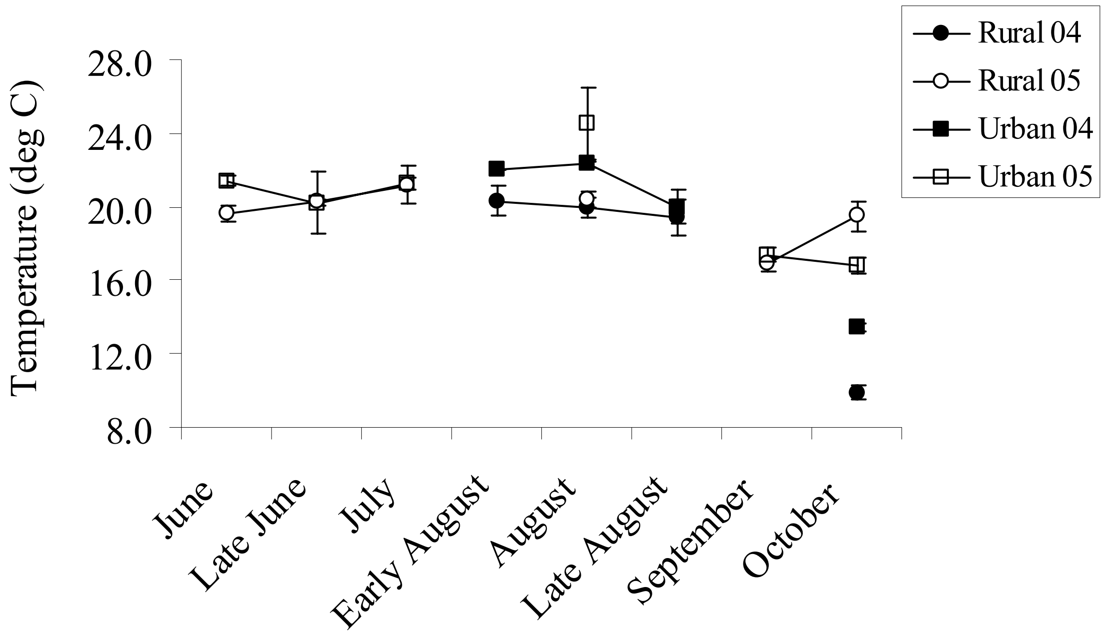


Figure 6. Plot of stream temperature (A) and dissolved oxygen (B) of urban and rural streams by year across the sampling season (2004-2005). Stream temperature (°C) was generally higher in urban streams than in rural streams, especially during the month of August. Dissolved oxygen (mg/L) was found to be lower in urban stream compared to rural stream sites. Differences in the year sampled were apparent. Data represent mean ± SEM bars.

(A)


Rural 04
Rural 05
Urban 04
Urban 05
28.0
24.0
20.0
16.0
12.0
8.0
Temperature (deg C)
June
Late June
July
Early August
August
Late August
September
October

(B)

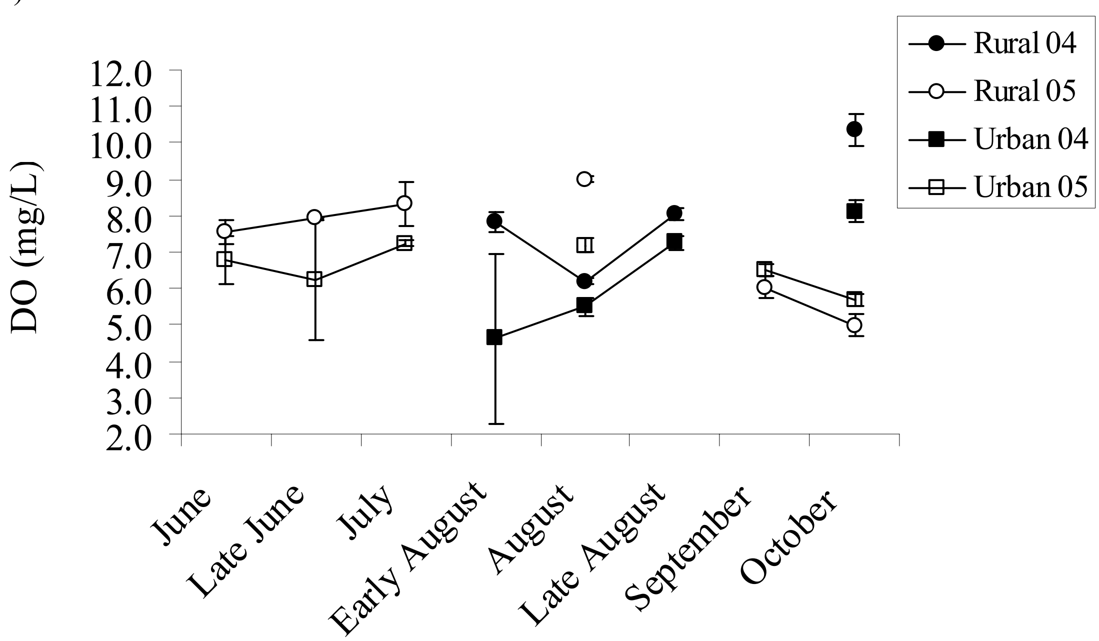
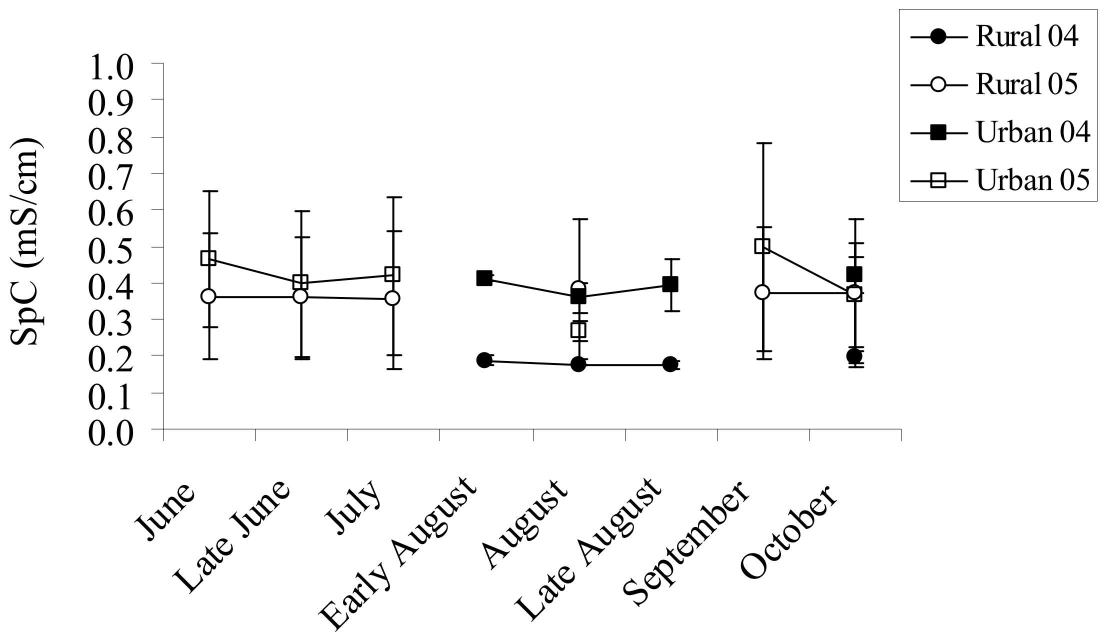

Rural 04
Rural 05
Urban 04
Urban 05
12.0
11.0
10.0
9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
DO (mg/L)
June
Late June
July
Early August
August
Late August
September
October

Figure 7. Plot of specific conductivity (A) and pH (B) of urban and rural streams by year across the sampling season. Urban sites displayed greater conductivity (mS/cm) than rural sites, however this pattern in pH is not to apparent. Data represent mean ± SEM bars.

(A)



(B)

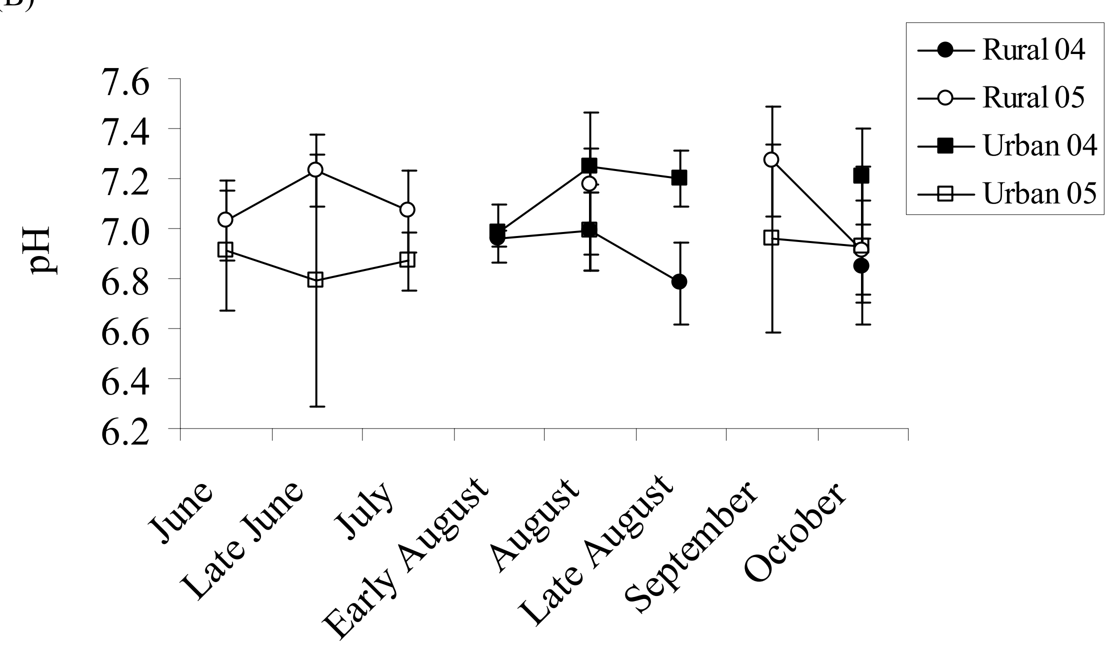


Figure 8. Continuous water temperature data for three of the stream sites sampled in 2005. Data was recorded using a single submerged Water Temp Pro logger (Onset Computer Corporation, Massachusetts, USA) at each site. One of the rural loggers was lost during the sampling season. Data loggers were initiated and positioned on June 25, 2005 and were subsequently removed from various locations between October 10 – 13, 2005. Water temperature was recorded in degrees Celsius (°C). *Rural: SENE; Urban: ANAC, CABJ*

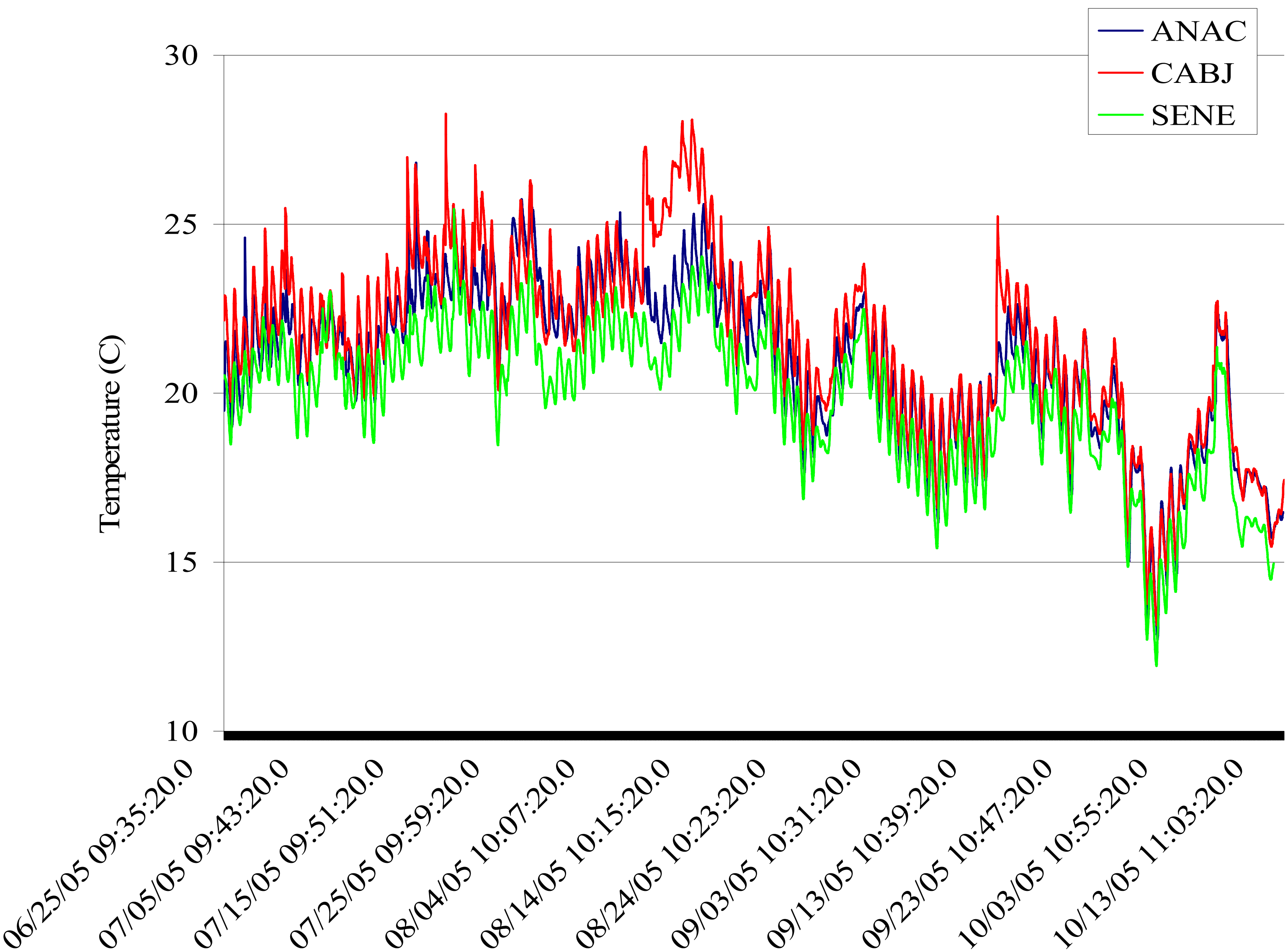

ANAC
CABJ
SENE
Temperature (C)
30
25
20
15
10
06/25/05 09:35:20.0
07/05/05 09:43:20.0
07/15/05 09:51:20.0
07/25/05 09:59:20.0
08/04/05 10:07:20.0
08/14/05 10:15:20.0
08/24/05 10:23:20.0
09/03/05 10:31:20.0
09/13/05 10:39:20.0
09/23/05 10:47:20.0
10/03/05 10:55:20.0
10/13/05 11:03:20.0

Figure 9. Population abundance of blacknose dace (BND) and creek chub (CKB) in urban and rural streams was estimated using the Jolly-Seber open population model. BND population abundance was greater than CKB at sites ($P < 0.05$), however differences between urban and rural populations within each species was not significant. Urban fish population abundance was slightly larger than rural populations when BND and CKB were combined ($P = 0.06$). Each bar indicates the average of two (2004) or three (2005) estimates of the population (± SEM).

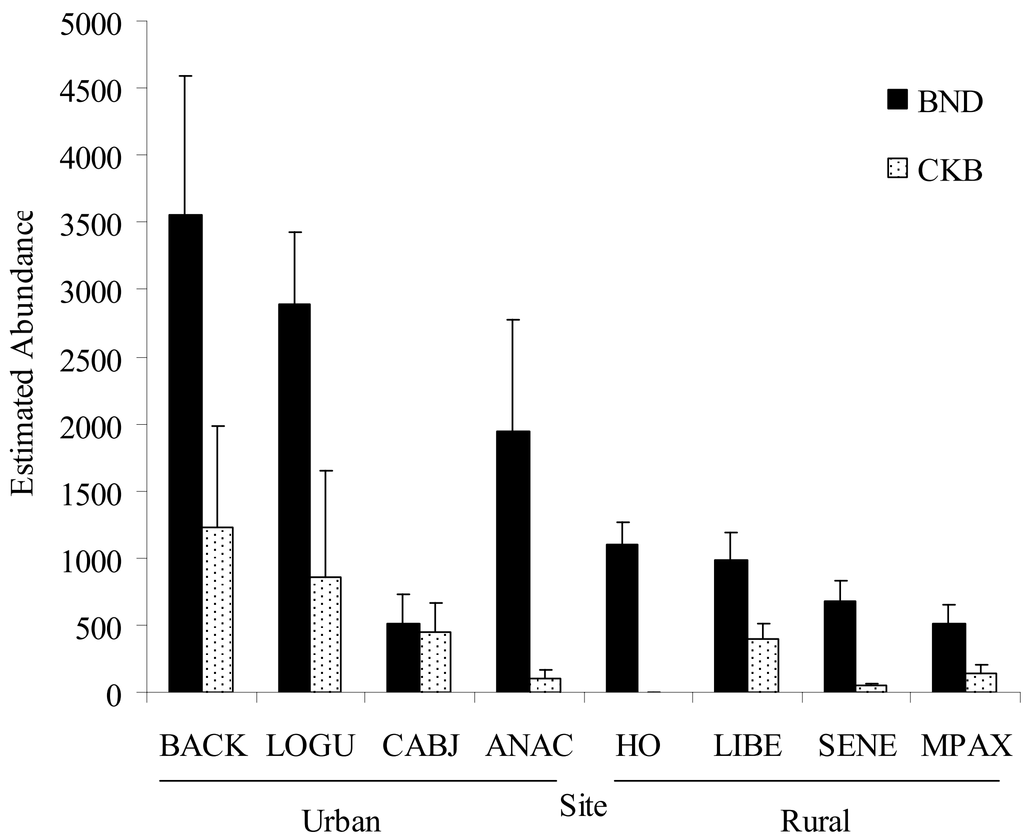


Figure 10. Proportion of movers and stayers in urban and rural streams.

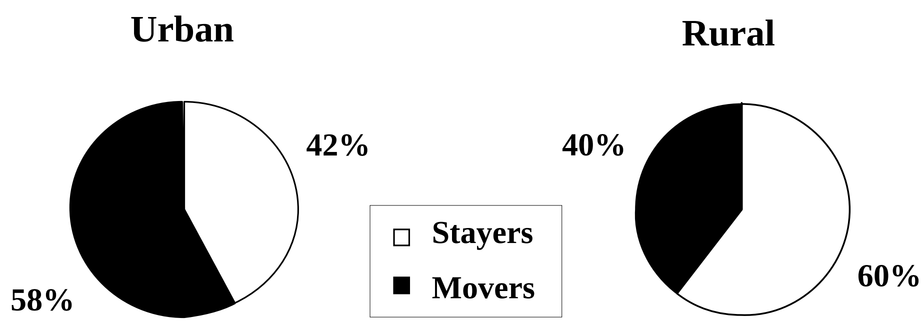


Figure 11. Unsigned movement of blacknose dace and creek chub in urban and rural streams.

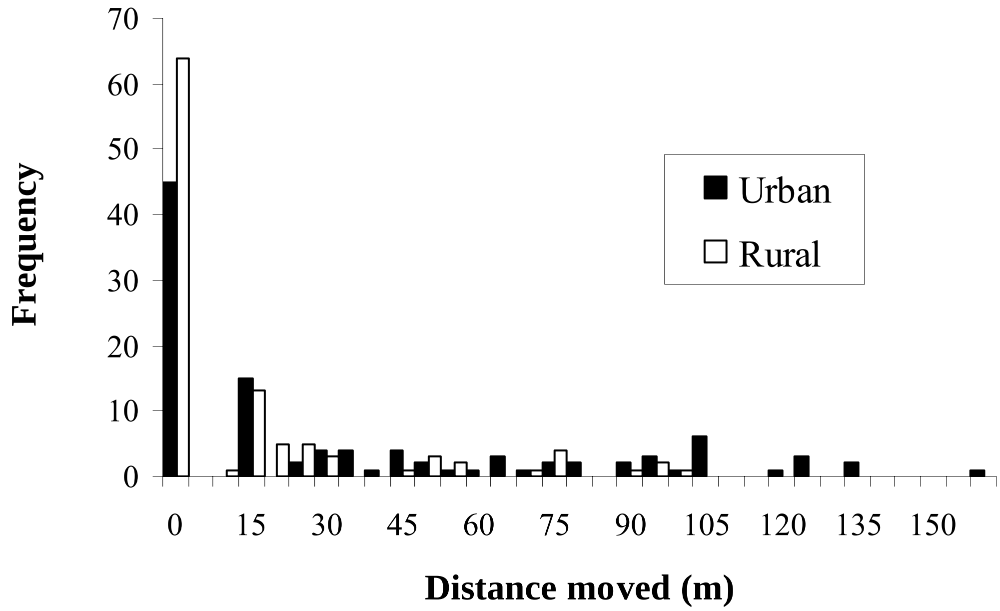
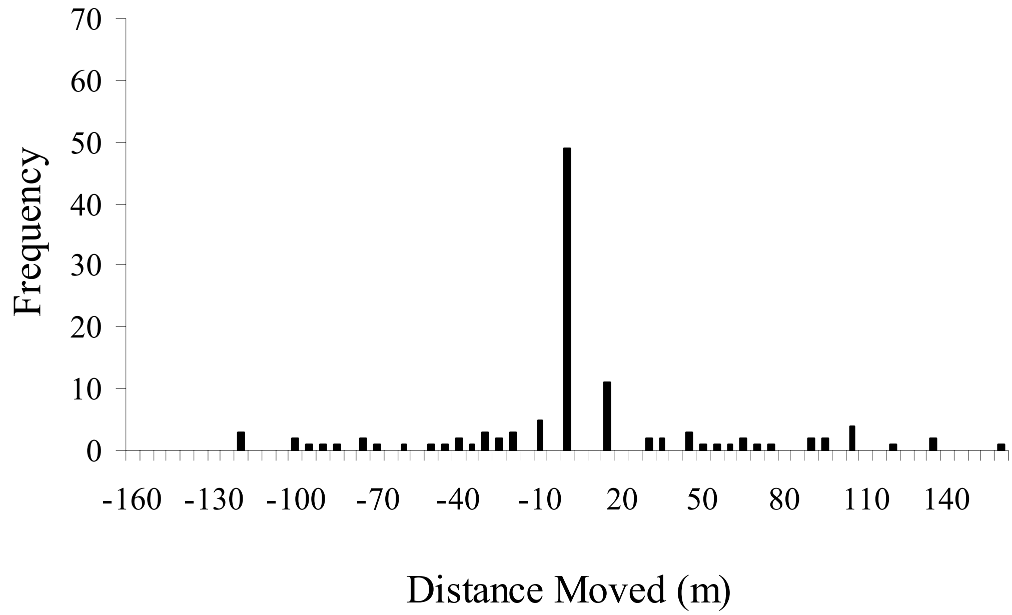


Figure 12. Signed movement of cyprinid populations in urban (A) and rural (B) streams.

A. Urban blacknose dace and creek chub.



B. Rural blacknose dace and creek chub.

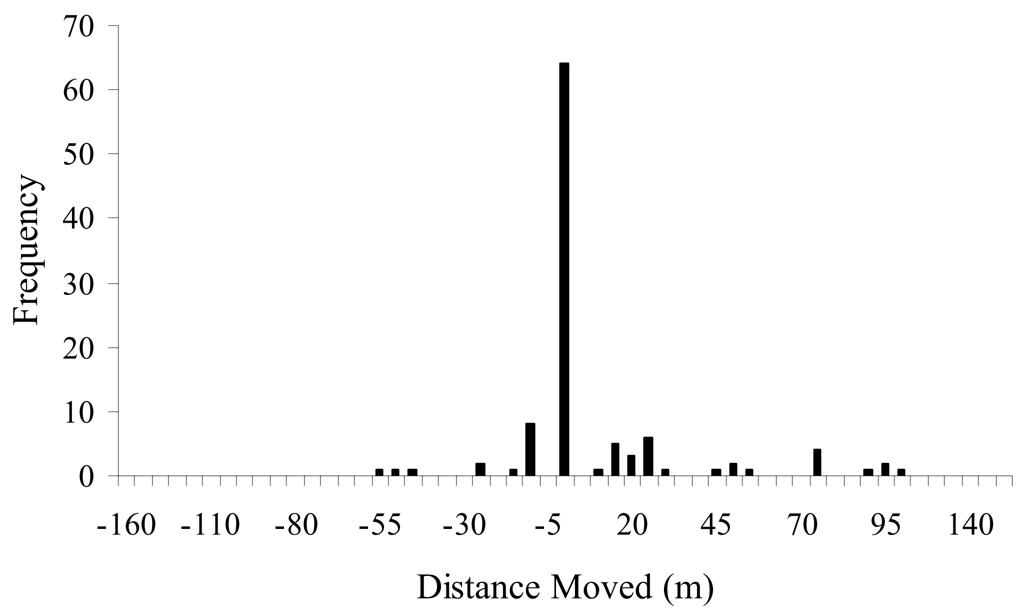


Figure 13. Relationship between total length and distance moved in urban mover blacknose dace.

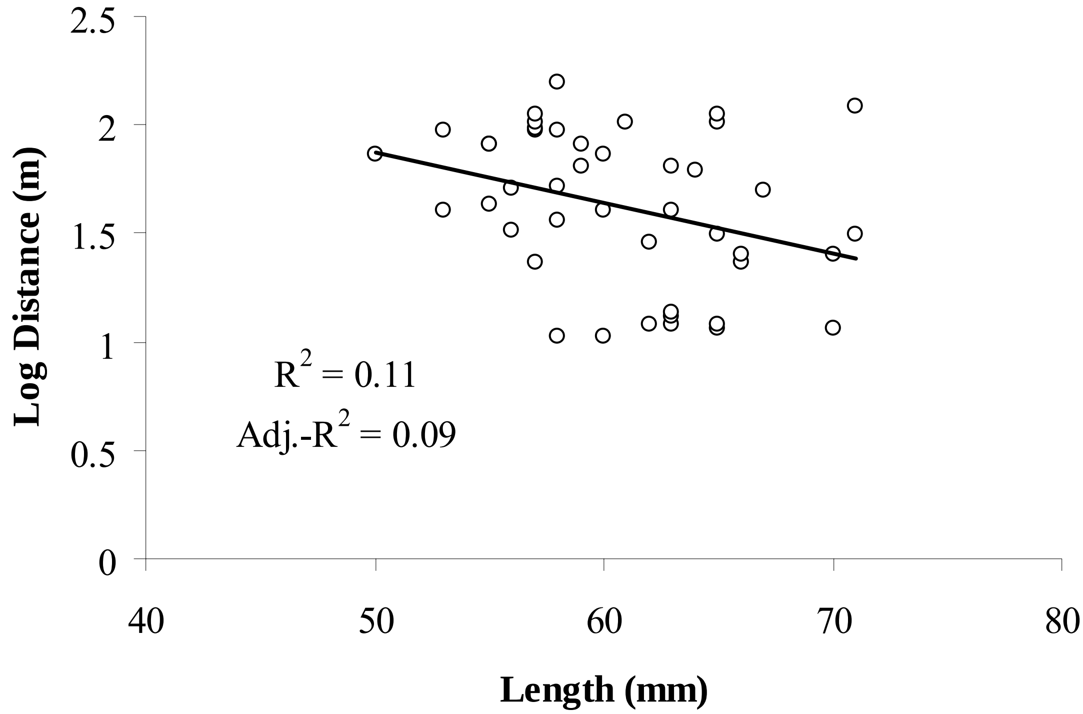


Figure 14. Relationship between total length and distance moved in urban mover creek chub.

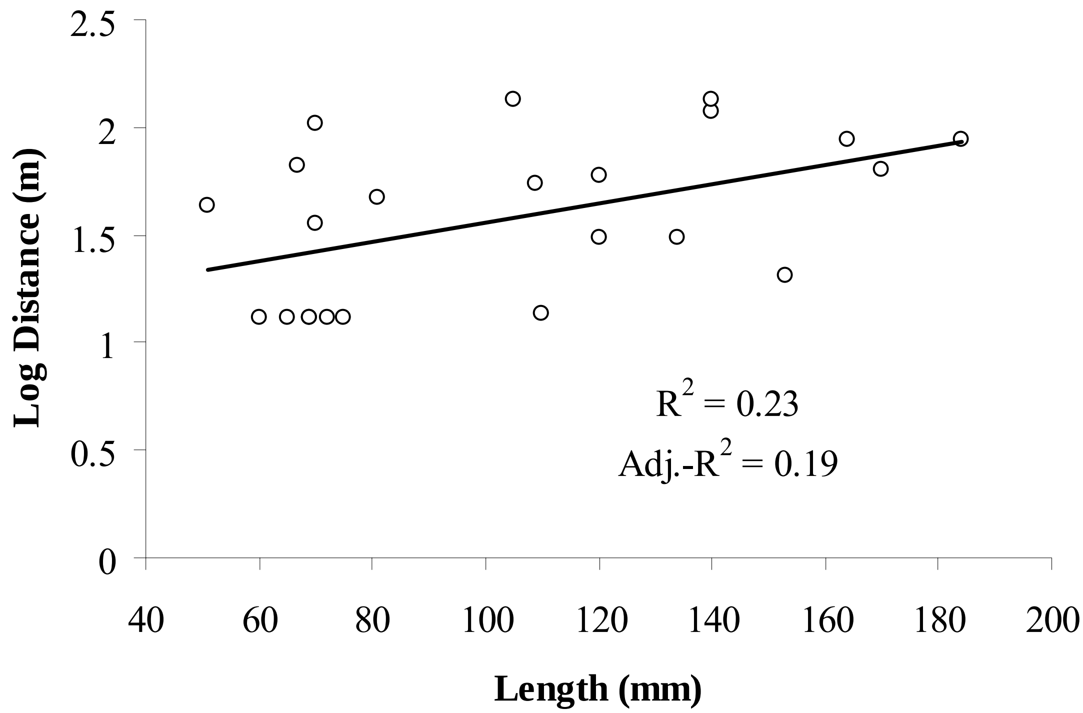


Figure 15. Average lengths of urban mover and stayer subpopulations. Creek chub are on average longer than BND. Length values used were those taken at the time of marking. (*) signifies a significant difference ($P < 0.01$) between movers and stayers.

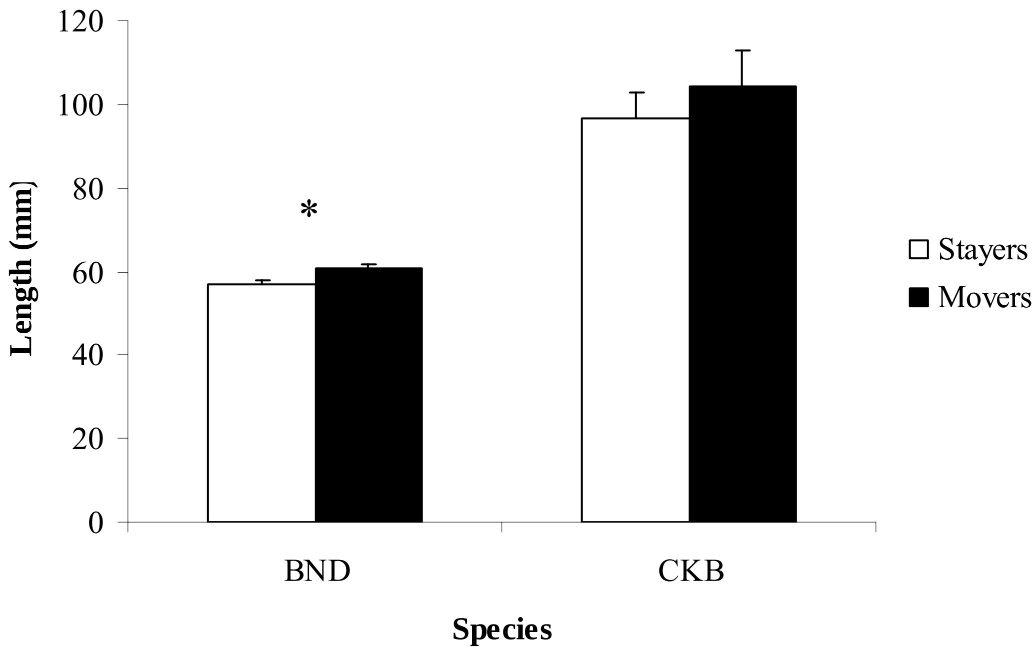
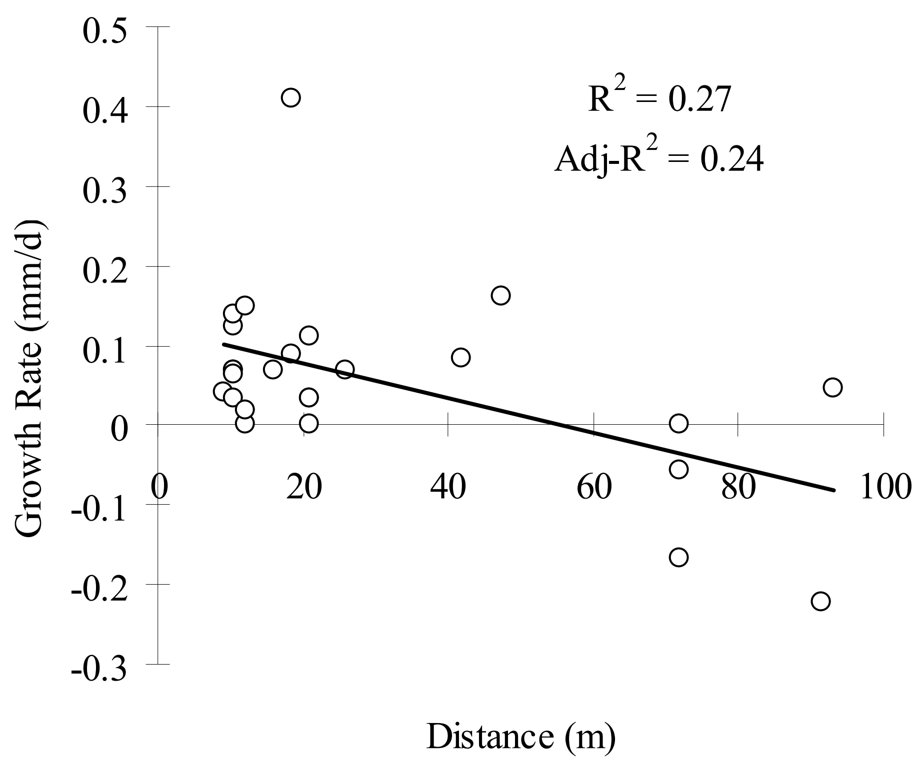


Figure 16. Effects of growth on distance moved by rural mover blacknose dace (BND). The more BND moved, the lower their daily growth rate. Growth rate (mm/d) was calculated by taking the difference of total length between captures and dividing by the number of days that passed between those dates. Since growth of the mover and stayer subpopulations were not different, growth was regressed on only the mover BND distance moved. (N = 25, $P < 0.01$)



Chapter 6: Conclusions

This work combines multiple approaches to understanding urbanization impacts on stream fish and their habitat. Through a database study and intensive fieldwork, I documented several important findings that add significant information to the already complex story of urbanization effects. My studies indicate that urbanization affects fish assemblage structure, channel habitat, fish habitat selection, and fish movement patterns.

Beginning with the general hypothesis that urbanization influences fish assemblages, I used the Maryland Biological Stream Survey data base to study changes in assemblage structure as urbanization intensity increases within a watershed. Surprisingly, the effects of urbanization were detected more easily in the eastern Piedmont than in the Coastal Plain physiographic province of Maryland. Urban land use (ULU) was found to significantly decrease species richness in eastern Piedmont streams, specifically in watersheds with greater than 25% ULU. The fish index of biotic integrity, a measure of fish assemblage health, declined significantly with percent urban land use. In addition, first and second order streams with > 25% ULU did not exhibit the expected fish assemblage, while in third order streams, the expected assemblage was found only in streams with < 50% ULU. This was the first study of urbanization effects in Maryland streams, and features results seen in other published works, broadening the geographic base of urbanization impacts on fish assemblages. It also served as a first step to understanding fish ecology in degraded streams.

The habitat complexity chapter presents some interesting findings, although some results were unexpected. While I was looking for more distinct changes in habitat characteristics along the urban-rural gradient than found, the lack of significant differences along the urban-rural gradient could be considered a significant finding. However, my data did point out a few important results. Stream conductivity increased linearly as urbanization increased, and was significantly higher in streams with > 30% ULU. In addition, the number of dewatered woody debris and the maximum height of erosion were higher in streams with > 45% ULU. These results indicate that channel habitat decreases with considerable suburban development, providing a degraded environment for stream biota. Finally, the increase in engineered banks in streams with > 60% ULU is an important finding because it indicates that these systems have already felt the influence of urbanization and someone is trying to alleviate these impacts. Although it is hard to accurately say where urbanization affects stream channels the most along the urban-rural gradient, there seems to be many breakpoints where specific changes could occur. This study did not measure all aspects of fish habitat, and thus may need further study to determine if thresholds exist for particular habitat parameters, or whether these changes occur in a more gradual, linear fashion.

The habitat patch selection study presented some key findings related to fish ecological responses to urbanization as well as biotic interactions in small stream communities. The similarity in fish habitat preference between rural and suburban stream populations was not surprising. However, the fact that the combined shade and large woody debris treatment was selected the most out of all the other treatments supports the basic tenet that complex habitat is superior to simple habitat. On the contrary, urban fish preferred the shade only treatment more than any other enhancement. Many urban channels are wide and lack shaded habitat, and therefore present the ideal environment to test the benefits of shade. These results demonstrate the need for intact riparian buffers to recruit large overhanging branches and minimal streambank erosion producing undercut banks to provide adequate habitat for fish in urban watersheds. In addition, the evidence for intraspecific competition among CKB of various sizes presents an interesting link between habitat quality, fish selection, and behavior. These small stream cyprinids have not been shown to occupy feeding positions previously, although my data suggests that the large individuals may interfere with juvenile habitat selection through a competitive hierarchy of dominant behavior.

Finally, results from the fish movement chapter propose that urban and rural BND and CKB populations have diverged in many respects. While my data did not support a significant difference in the proportion of movers and stayers in urban and rural streams, urban fish do select and occupy a larger expanse of stream pool habitat than rural fish. This is a significant finding for many reasons. Biological monitoring and population estimates are used to detect changes in stream biota. If a stream population is monitored on a yearly basis and extreme differences in fish abundance are found potentially due to a largely mobile population, conclusions that a population is suffering may be made incorrectly. Additionally, these results are crucial to the success of watershed restoration and habitat rehabilitation of biotic communities. Would fish return to and/or stay in a habitat patch that has been restored, particularly in urban stream channels?

The reasons for movement patterns in small stream cyprinids could be multiple; however, ecological interactions are likely a leading cause. Competition between BND and CKB, as well as among each population may play an important role in pool selection and therefore the size of habitat fishes use. Interestingly, this is analogous to conclusions drawn from the previous chapter and habitat complexity was shown to be reduced in urban stream channels in Chapter 3. Since urban fish assemblages in small streams are so simplified, composed of sometimes only two or three species, the foraging resource base and favorable habitat may be limited due to high niche overlap compared to rural streams. This creates the need to frequently disperse throughout the stream reach to meet the ecological, energetic, and behavioral demands of survival in a degraded environment. By traveling from pool to pool, fish can assess the potential and actual energy input, and subsequent energy output from swimming to another pool, as well as the risk taken with other individuals present. Conceptually, this applies to any fish, and provides a framework for movement patterns in both urban and rural stream populations.

In conclusion, research on urban fish populations presented many more interesting findings than originally hypothesized. It is my hope that this work will aid future land and fisheries managers to understand not only that fish assemblages are impacted by urbanization, but how they respond to degradation of stream habitat and the surrounding environment. After spending considerable time in urban streams, it was a pleasure to inform curious minds along the streambank that there were in fact fish living in their city stream. Although the fish species I chose were pollution tolerant and therefore fitting to conduct these studies with, they are not tolerant to all environmental stressors. BND and CKB are the pioneers of small, first order streams and therefore must continue to adapt to survive in these harsh environments if land use change continues at the current rate.

There is still much to learn from this area of research. As stated earlier, habitat degradation needs to be better understood along the urban-rural gradient since my study was not able to pinpoint breakpoints or gradual change in many habitat parameters. This would be beneficial to guide land managers as to how much construction and development could occur before significant changes emerged in the stream ecosystem. The prevalence of stream restoration practices indicates that 1) we are changing a significant portion of stream and watershed processes and 2) it is important to do something about it. Few stream restoration projects have determined if stream biota respond to habitat enhancements. In mimicking a short-term response to habitat enhancement, I only scratched the surface of potential research on this topic. Further study of long-term responses to stream restoration practices providing evidence of which restoration techniques work and which ones do not would help secure future grant support and project monies. In this case, I would suggest that a BACI (before-after-control-impact) design be conducted to provide comprehensive picture of how restoration has benefited the stream community. Finally, future studies on fish movement patterns in restored and unrestored urban stream systems would provide critical information regarding whether fish communities not only use restored habitat, but how long they occupy it, and for what reasons. By offering fish an increased foraging base in addition to streambank stabilization or other restoration practices, one could determine if forage or other habitat parameters were the reason for habitat patch selection.

Appendices

Appendix I. Eigenvector weightings for two principal components that related impervious surface and urban land use to physical habitat attributes.

PC 1	Eigenvectors	PC 2	Eigenvectors
Urban	0.46	Urban	0.37
Impervious surface	0.45	Impervious surface	0.42
Maximum depth	0.48	Rootwads	0.49
Pool quality	0.38	Woody debris	0.31

Appendix II. Treatment structure of the patch selection experiments. The number of sites (replications) that were sampled in each treatment and land use category are represented below. Each treatment was paired with a control habitat patch to provide enhanced and un-enhanced habitat qualities from which the fish select (*Both = shade and woody debris, ULU = urban land use*).

	Shade	**Woody Debris**	**Both**
Rural (< 15% ULU)	4	4	4
Suburban (27 – 46% ULU)	4	4	4
Urban (> 60% ULU)	4	4	4

Appendix III. Dimensions of LWD used in the woody debris habitat enhancement experiment.

Log	Length (m)	Diameter (m)	Volume (m^3)
A	2.04	0.10	0.016
B	1.50	0.12	0.017
C	0.90	0.18	0.023

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