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**THE RESPONSE OF FIRST AND SECOND ORDER STREAMS TO URBAN
LAND-USE IN MAINE, U.S.A.**

By

Chandler C. Morse

B.S. The Ohio State University, 1996

A THESIS

Submitted in Partial Fulfillment of the

Requirements for the Degree of

Master of Science

(in Ecology and Environmental Science)

The Graduate School

The University of Maine

May, 2001

Advisory Committee:

Christopher Cronan, Professor of Botany and Ecology, Co-advisor

Alexander D. Huryn, Associate Professor of Aquatic Entomology, Co-advisor

Chet Rock, Professor of Environmental Engineering

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By Chandler C. Morse

Thesis Co-advisors: Dr. Christopher Cronan and Dr. Alexander D. Huryn

An Abstract of the Thesis Presented
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Degree of Master of Science
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Physical, chemical, and biological characteristics of streams draining 20 catchments in Maine, U.S.A were compared to determine the influence of increasing urban intensity on stream ecosystem structure. The catchments had varying levels of urban land-use (percentage of the total impervious area within the catchment) ranging from 1-31%.

Stream habitat quality, stability, and water quality consistently decreased as the proportion of impervious surface area increased within the catchment. Indices based on stream benthic macroinvertebrate communities showed even stronger declines as a function of increasing impervious area in the study catchments.

Streams draining catchments with levels of impervious surfaces <6% had higher levels of both total and Ephemeroptera + Plecoptera + Trichoptera (EPT) taxonomic richness. With increased levels of urban intensity, benthic macroinvertebrate communities in streams were characterized by decreased numbers of sensitive taxa. Taxa considered to be moderately sensitive to anthropogenic stress (e.g. *Acerpenna*

(Ephemeroptera), *Paracapnia* and *Allocapnia* (Plecoptera), *Optioservus* and *Stenelmis* (Coleoptera), *Hydropsyche* and *Cheumatopsyche* (Trichoptera), Orthocladiinae (Diptera), and Oligochaeta) were apparently little influenced by increasing urban intensity. These patterns were similar between the Fall and Spring.

Results indicated that beyond an apparent threshold of ~6% impervious surface area in the catchment, study streams exhibited an abrupt step-like drop in macroinvertebrate community condition as indicated by a reduction in the presence of sensitive macroinvertebrate taxa. Streams with <6% impervious surfaces contained invertebrate communities with average total richness of 33 taxa (Fall) and 31 taxa (Spring) and average EPT richness of 15 taxa (Fall) and 13 taxa (Spring). In contrast, none of the streams located in catchments with 6-27% impervious coverage exhibited average total richness >18 taxa and average EPT richness >6 taxa. Physical habitat and water quality parameters failed to indicate the mechanism resulting in degradation of the macroinvertebrate community.

DEDICATION

This thesis is dedicated in loving memory of Miriam Christie Higgins (1910-2000) and Catherine Murphy Morse (1917-1998).

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PREFACE: THESIS STATEMENT

Hynes (1975), in his now classic paper entitled "The stream and its valley," described the many ways in which the ecological attributes of a stream are controlled by characteristics of its watershed. Although the main focus of Hynes' paper was on the relationship between streams and their riparian corridors and surrounding forests, he stated clearly that "changes in the valley wrought by man may have large effects" (Hynes 1975, p. 12).

As we begin the 21st century, there is concern that streams are increasingly at risk from urbanization. McDonnell & Pickett (1990) defined urbanization as "an increase in human habitation, coupled with increased per capita energy consumption and extensive modification of the landscape". Urban land-use occupies 26 million hectares of land in the U.S.A. (Vesterby 1994), and the total area of urbanization is expected to increase rapidly within the next few decades. The most rapid growth is occurring in new suburbs located 15-40 km outside of metropolitan areas (O'Hara 1997). These suburbs are primarily composed of low-density residential neighborhoods, as opposed to the traditional high-density urban development commonly associated with metropolitan areas. Such suburban "sprawl", defined as "the lack of continuity of expansion" (Peiser 1989), is resulting in both a rapid increase in the quantity of natural area that is being influenced by urbanization as well as an increase in the range of urban intensities by which streams are being affected.

Urban land-use has been associated with a complex set of negative influences on major components of stream ecosystems (Fig. 1). The hydrologic regime of urban

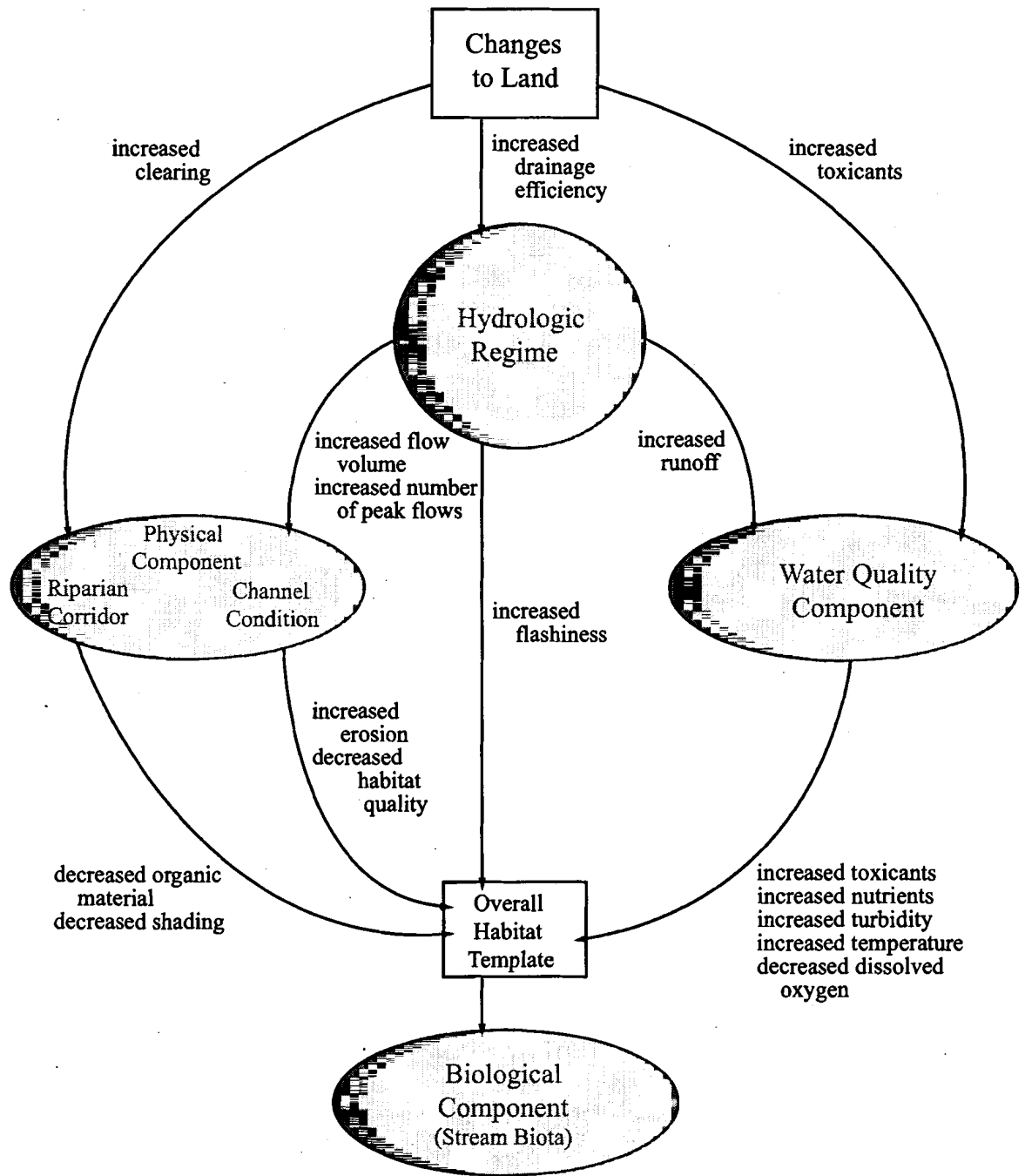


Figure 1. Conceptual model of the direct and indirect influences of urbanization on stream attributes (gray shading denotes topics of review in Chapter 1).

streams is characterized by larger storm flows rising to peak at a much faster rate and occurring with a higher frequency (Hollis 1975, Ragan et al. 1977, Booth 1991). These influences, along with direct effects from urban development such as riparian clearing, drive physical changes within the stream channel such as widening and incision as well as increased erosion and sedimentation (Leopold 1968, Hammer 1972, Whipple et al. 1981, Arnold et al. 1982, Booth 1990). Urban land-use also results in degraded water quality as urban runoff often has elevated levels of organic compounds, suspended and dissolved solids, nutrients (nitrogen and phosphorous), and heavy metals (Porcella & Sorenson 1980). These physical and chemical effects of urbanization are often correlated with changes in the overall habitat template within the stream as reflected by alterations to the community composition of stream macroinvertebrates (Table 1).

Recent research concerning the nature and extent of degradation of stream ecosystems due to urbanization has resulted in a number of patterns and areas for further research. Schueler (1994) reviewed 18 studies investigating the relationship between urban land-use and stream ecosystem quality and concluded that the percentage of the total impervious area (PTIA) within a stream's catchment may serve as a predictor of aquatic system impairment. The author reported that stream health, as indicated by changes in stream hydrology, habitat, water quality, and biodiversity, consistently showed degradation at PTIA levels between 10-20%, suggesting an important threshold effect. He also suggested the need for further research in varying locations, including the northeastern U.S.A., to assess the utility of this potentially valuable tool for aquatic resource conservation.

Table 1. A review of selected studies of the effect of urbanization on the stream invertebrate communities including the location of the study and the apparent threshold of PTIA beyond which degradation was apparent. Nr indicates data not reported.

Author	PTIA		Results
	State	Threshold	
May (1997)	WA	5-10%	decreased multimetric summary with increased urban intensity suggests altered hydrologic regime and riparian clearing as causative factors
Klein (1979)	MD	10%	decreased diversity with increased urban intensity
Shaver et al. (1995)	DE	8-15%	decreased diversity with increased urban intensity notes correlation between community degradation and habitat quality
Schueler & Galli (1992) ¹	MD	15%	decreased diversity with increased urban intensity
Maxted (1996)	DE	10-15%	loss of sensitive taxa with increased urban intensity shift in community towards tolerant taxa (chironomids) notes correlation between community degradation and habitat quality
Jones & Clark (1987)	VA	15-25% ²	decreased diversity with increased urban intensity shift in relative abundance of tolerant taxa (chironomids)
Duda et al. (1982)	NC	nr	decreased diversity and loss of sensitive taxa with increased urban intensity shift in community towards tolerant taxa (oligochaetes and chironomids) suggests toxic substances and organic wastes as causative factors
Pedersen & Perkins (1986)	WA	nr	decreased functional diversity with increased urban intensity shift in community towards taxa tolerant of erosional/depositional habitat and low food quality
Benke et al. (1981)	GA	nr	decreased diversity with increased urban intensity
Garie & MacIntosh (1986)	NJ	nr	decreased population density with increased urban intensity decreased richness and loss of sensitive taxa with increased urban intensity shift in dominance towards tolerant taxa (tubificids and chironomids) notes consistent trends with community degradation and metal concentrations
Whiting & Clifford (1983)	AB, Canada	nr	increased population density with increased urban intensity decreased diversity and loss of sensitive taxa with increased urban intensity shift in community towards tolerant taxa (tubificids and chironomids) suggests nutrient loading and sedimentation as causative factors
Pratt et al. (1981)	MA	nr	decreased diversity with increased urban intensity loss of sensitive taxa with increased runoff influence suggests influence of urban runoff as causative factor

¹ Schueler & Galli (1992) as found in Schueler (1994).

² Conversion of Jones & Clark (1987) urban intensity to a PTIA value from Schueler (1994).

This study attempted to answer two questions. First, is the level of degradation of the physical, chemical, and biological condition of 1st and 2nd order streams in Maine related to increasing urban intensity across an urban gradient from low to high PTIA? Second, do adverse impacts of urbanization on Maine streams occur at a threshold of 10-20% PTIA as suggested by Schueler (1994)? These questions were answered by comparing physical, water quality, and biological characteristics from 20 1st and 2nd order catchments with levels of urban intensity ranging from ~0% to 31% PTIA. In the chapter that follows, a literature review is provided concerning the degradation of streams as a result of urban development. In Chapter 2, the methods and results of this field study are presented and discussed.

CHAPTER 1
BACKGROUND LITERATURE REVIEW: URBANIZATION
AND STREAM DEGRADATION

Introduction

May (1996) defined urbanization as “the modification of a landscape from a rural, agricultural, or forested condition to one dominated by residential, commercial, and/or industrial land-use”. This modification in the landscape results in direct and indirect influence that lead to changes in the major components of stream ecosystems; including the hydrologic regime, channel form, water quality, and the biological community. This review investigates these changes in detail as well as the emergence of a cumulative indicator of the level of degradation associated with urbanization.

Influence on Hydrologic Regime

Urban land-use has major effects on the hydrological regime of streams and it is through these effects that the most significant ecological changes occur (McGriff 1972, Schueler 1987, Arnold & Gibbons 1996). Changes from rural to urban land-use alter stream hydrologic regimes by affecting the infiltration and runoff patterns and the amount of evapotranspiration (May 1996, Poff et al. 1997). In general, hydrological changes due to urbanization include larger peak flows following storms, a decrease in the time for these peak flows to occur, an increase in the number of storms that result in bankfull conditions, and a greater frequency of periods with intermittent surface flow (Leopold 1968, Hollis 1975, Arnold & Gibbons 1996). Also, it is hypothesized that urbanization

may result in an overall decrease in the baseflow of streams (Simons & Reynolds 1982, Evett 1994). Finally, the alteration of the thermal characteristics of the local atmosphere and the “urban heat island effect” can increase the level of precipitation and the intensity of storm events associated with urbanized landscapes (Pope 1980).

Altered Runoff Regime

Streams of the northeastern U.S.A. are characterized by hydrologic regimes that are dominated by subsurface flow. The actual patterns of discharge, however, are determined by a suite of within-catchment characteristics including the amount of rainfall and the frequency and intensity of storms, in addition to overall catchment gradient, and surficial and bedrock geology (McGriff 1972, Poff et al. 1997). In general, the surficial geology of southern and central Maine is dominated by glacial till composed of a heterogeneous mixture of sand, silt, clay, and stones and glaciomarine deposits composed primarily of silt, clay, and sand (Thompson 1985). This till is usually covered by a layer of permeable topsoil and organic detritus. This results in soils that generally permit efficient infiltration of rainfall, which is transported to the stream as interflow (Dunne et al. 1975). Overland flow occurs only after rainfall sufficient to saturate soils and raise the level of the watertable so that it intercepts the soil surface. The United States Environmental Protection Agency (US EPA 1993) concluded that, in the northeast, 10% of the precipitation results in direct and overland runoff, whereas 50% is associated with infiltration to groundwater supplies (the remaining 40% leaves the catchment as evapotranspiration).

Urbanization influences patterns of runoff by altering the overall catchment gradient through grading, decreasing the amount of surface vegetation, increasing soil compaction, reducing soil storage capacity, and increasing the proportion of the land surface covered with impervious surfaces which prevent infiltration (May 1996). The most significant of these changes is believed to be the increase in the proportion of total impervious area (PTIA) within the catchment (Schueler 1987, 1994). Hard, smooth surfaces simply reduce infiltration, resulting in rapid runoff over the land surface towards streams and ponds (Horton 1975, Poff et al. 1997).

Increased Runoff Volume

The runoff coefficient (R_v) is the proportion of rainfall collected by surfaces that contribute runoff. An average parking lot, for example, has an R_v of 0.95 as opposed to 0.06 for an unpaved meadow (Schueler 1994). On the basis of these values, Schueler (1994) suggested that a storm producing 2.5 cm of precipitation on a hectare of paved lot would produce $\sim 2.3 \times 10^5$ L of runoff. A meadow of the same size would produce only $\sim 2.5 \times 10^4$ L. Similarly, the US EPA (1993) suggests that catchments with 30-35% PTIA will have 30% of precipitation resulting in direct and overland runoff and only 35% of precipitation infiltrating through the soil to the groundwater supply.

Observations indicating that the magnitude of runoff increases with urban land-use support these estimates. A study in South Africa comparing a rural and a suburban stream in Johannesburg found that runoff accounted for 4% and 15% of precipitation, respectively (Stephenson 1994). In their review, Arnold & Gibbons (1996) summarized findings suggesting that with 10-20% PTIA, the amount of runoff is double that of a

similar non-urban catchment. As PTIA rises to 30-50%, the amount of runoff volume can triple (Arnold & Gibbons 1996). Codner et al. (1988) supports these findings with results from a paired catchment study in Canberra, Australia. They found that a low-density urban stream compared to a rural stream received six times more runoff. In a study of East Meadow Brook on Long Island, Seaburn (1969) found that the increase in runoff volume was proportional to increases in development. Runoff increased by a factor of three as development increased by a factor of five. This increase in the volume of runoff, as a consequence of urbanization, can also be exacerbated by the construction of storm sewers. Leopold (1968) showed that increases in the amount of runoff reaching the stream differed greatly within areas of the same urban intensity depending on attributes of storm sewer construction.

Decreased Lag Time

Along with an increase in the catchment area that prevents infiltration and provides a smooth surface facilitating runoff, urbanization is associated with a dramatic increase in drainage density. Drainage density refers to the area of drainage channels facilitating precipitation reaching the stream that are present within the catchment. Within urbanizing catchments, drainage density is increased by the construction of runoff conveyance systems such as storm sewers. Graf (1977), in a study of a suburban stream in Iowa City, found that a 20% increase in the level of urbanization over 75 years doubled the drainage density due to the proliferation of storm sewers in residential areas. He also found a strong correlation between development of runoff conveyance systems

and a decrease in lag time, or the time between the peak of the storm and the peak in stream discharge.

Lag time is a function of basin slope and basin length (McGriff 1972). Grading and increases in PTIA combine to increase the effective basin slope. The construction of roadways and storm sewer drainage systems combine to reduce effective basin length. Peak flows following storms can occur as much as 50% faster in urban conditions as opposed to similar non-urban settings (Schueler 1987). Codner et al. (1988) noted in their study in Australia that the lag time of an urban stream in the study was 30 times shorter than a non-urban stream.

Increased Peak Flow

The hydrograph of urban streams is characterized by a sharply angled rising limb with peak discharges higher than those in the absence of urbanization (Fig. 2, Leopold 1968). Ragan et al. (1977), on the basis of work in the Anacostia watershed in Maryland, reported that storms occurring with a 1.1 year frequency showed a 49% increase in peak flow following heavy urban development. They also noted that peak flows of 2, 5, 10, and 20-year floods increased 38%, 31%, 24%, and 23% respectively (Ragan et al. 1977). Seaburn (1969), in his study of a Long Island, New York stream, recorded an increase in average peak flow from $8.9 \times 10^3 \text{ L s}^{-1}$ in pre-urban conditions to $2.2 \times 10^4 \text{ L s}^{-1}$ following urban development within the catchment. On the basis of a review, Leopold (1968) reported PTIA equaling 20%, 40%, and 60% will increase peak flow by a factor of 1.5, 2.7, and 4.3, respectively. The magnitude of such an increase will be a function of the type as well as the proportion of impervious area (Hollis 1975).

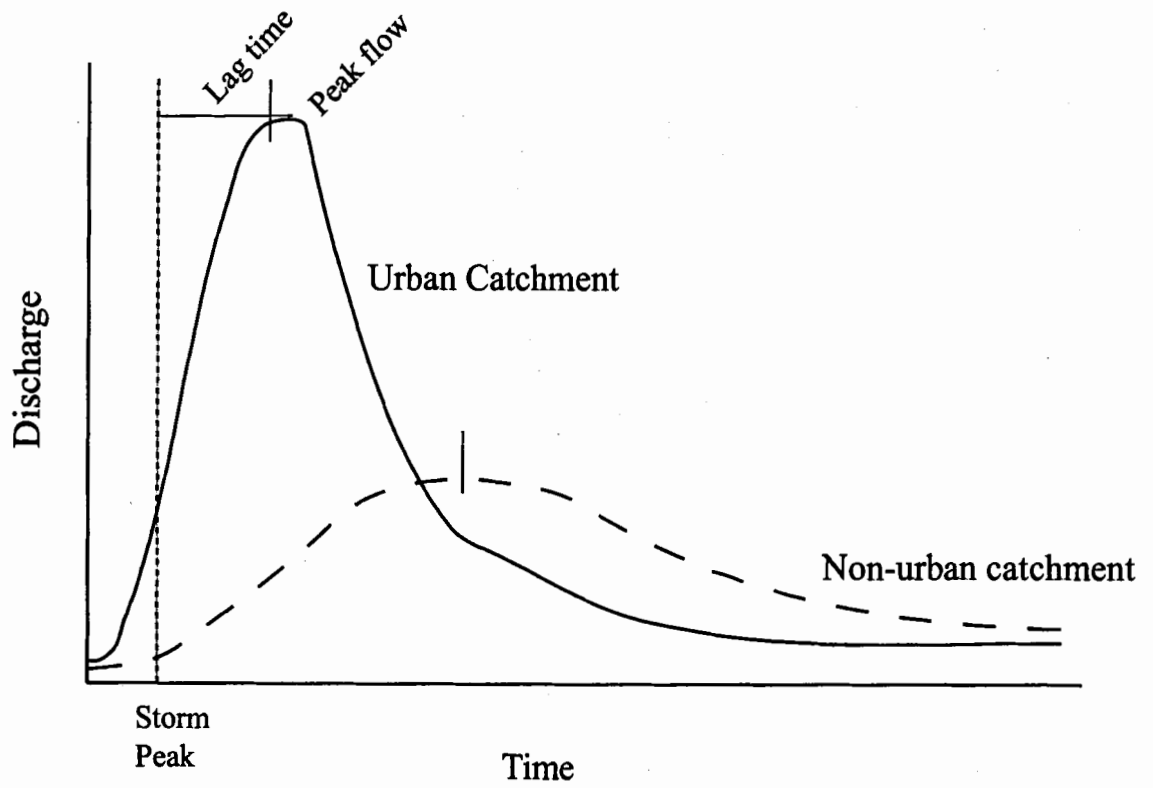


Figure 2. Hypothetical hydrograph of an urban- and non-urban-influenced discharge event (modified from Leopold 1968).

The relationship between increases in peak flow and urbanization is clear. However, there is considerable variability among reports concerning the extent of such increases. Hollis (1975) warned against forming a rule of thumb for predicting peak flow increases, noting that influences on the hydrologic regime will be site-dependent due to the effects of geology, geomorphology, catchment size, and regional climate. Hollis (1975) also noted that proportional increases in peak flows are greater for smaller storms. This is also evident in the research of Ragan et al. (1977), as noted above. Infrequent but large storms may saturate the soils, masking the additional influence of impervious surfaces. Hollis (1975) also noted the possibility that the conveyance system of runoff can reach their maximum flow volumes during large storms, causing upstream flooding within the catchment and actually maximizing the peak flow at relatively lower discharge volumes within the stream itself.

Increased Number of Peak Flows

Urbanization has been associated with an increase in the number of storms that result in bankfull flows. Bankfull flows are commonly associated with storms having 1-2 year return intervals under reference conditions (Booth 1990). Booth (1990) noted that these bank-full flows instead may occur as many as 3-4 times per year in heavily urbanized areas. Similarly, Schueler (1987) noted that streams that normally would have reached bankfull levels once every two years under reference conditions can reach them 3-4 times per year due to storm sewer development. This increase in bankfull flows is due to a reduction in the quantitative role of infiltration in the catchment water budget.

Soil layers that are not affected by urbanization may actually store large amounts of water that reach the stream over a longer duration or not at all.

Decreased Baseflow

Urbanization is suspected to influence patterns of the baseflow of streams, although few studies have documented this phenomenon and none have conclusively identified the cause. Simons & Reynolds (1982) studied six streams on Long Island, New York with varying land-use and reported that urbanization reduced the baseflow of streams by as much as 60%. It was suggested that this was because of a decrease in groundwater supply due to a reduction of the surface infiltration capacity (Simons & Reynolds 1982). A similar result was reported for another study of 16 North Carolina streams (Evetts 1994). Additionally, Ferguson & Suckling (1990) and Harbor (1994) found that urbanization results in a greater frequency of intermittent streams.

Urbanization and Stream Channel Form

Stream channel form results from a dynamic equilibrium between the hydrologic regime and sediment supply (Carling 1988). Channel size is determined by the frequency of discharges capable of moving significant amounts of sediment within the channel (Booth 1990). Levels of discharge that fill at least at least 60% of the stream channel capacity are suspected to be the most significant in maintaining channel form. Smaller discharges may “winnow” fine sediments from the bank material (Carling 1988). Channel-forming discharges often have 1.5-2 year recurrence intervals (Leopold 1968) and occur with higher magnitude and frequency under urban conditions compared with

reference conditions as previously noted. The readjustment of the stream channel to accommodate increased peak discharge levels that occur more frequently includes channel enlargement, increased bank erosion, altered patterns of bed scour and fill, and increases in large-scale bank failures. These changes generally result in an alteration of stream habitat.

Channel Alterations

Rutherford & Ducatel (1994) described the basic process of channel re-equilibration in response to catchment urbanization. During the initial phases of urban development (e.g. the construction of the urban infrastructure), there is an increase in basin-derived sediments eroded from denuded surfaces. These accumulate in the stream channel. Sediment loads during this phase can be 2.5×10^2 - 4.0×10^4 times greater than loads characterizing non-urbanizing catchments (McGriff 1972). As urbanization proceeds, sewers may be installed. This results in an increase in the runoff volume. Hence, the stream competence, or the ability of the stream to move sediment, will increase and the stream channel will widen or deepen or both. Poff et al. (1997) noted that it can take streams centuries to regain equilibrium from alterations to the hydrologic regime and may actually remain in a constant state of recovery. Leopold (1972) echoed these findings in his long-term survey of Watts Branch near Rockville, Maryland where he found a reduction in the channel area with increased development until an apparent discharge threshold was reached. This threshold occurred when increases in discharge became larger than channel capacity. Once this threshold was exceeded, dramatic increases in channel area occurred (Leopold 1972). Large-scale bank failures and mass

wasting events can further confound the rate at which the channel re-equilibrates (Booth 1990).

Additional observational studies reinforce the conclusion that urbanization results in channel enlargement, while providing insight into its nature and extent. Hammer (1972), in a survey of 78 streams in Pennsylvania, found that channel width was proportional to the extent of urban development, with channels widening up 3.8 times that of reference conditions. He found the extent of enlargement to be influenced by the nature of PTIA and drainage efficiency, with greater increases in channel area associated with large and connected levels of impervious catchment area and, as previously noted, the presence of storm sewers (Hammer 1972). In addition, channel enlargement was not as prevalent in relatively newer or older urban areas (urban areas <4 or >30 years old; Hammer 1972). In a five-year study of Pheasant Branch in Middleton, Wisconsin, Krug & Goddard (1986) noted a 35% increase in width of streams draining developed catchments. Neller (1988), in a comparison of a rural and urban stream in Armdale, Australia, noted an increase in channel width by a factor of 2.4 for the urban stream five years after development. Robinson (1976), in an evaluation of eight streams in Maryland, found that urban streams had twice the channel area and their width to depths were 1.7 times greater.

Booth (1989) modeled the response of stream channels to increased peak flows caused by up to 10% PTIA and found that channel area may increase as much as 75%. Booth (1989) also noted that under these conditions, channel enlargement could result in incision that is disproportional to that predicted on the basis of the altered hydrologic regime alone. This disproportional incision was called "catastrophic" by Booth as it

results in deep, heavily eroded channels with steep banks that are predisposed towards mass wasting and cave-ins with the potential of damaging stream-side houses or roadways (Booth 1989). A stream's potential for incision primarily is a function of channel slope and streambed geomorphology, but can also be influenced by characteristics of flow, topography, geology, and channel roughness (Booth 1989).

Increased Erosion

Erosion rates are a function of catchment vegetation, soil type, slope, and patterns of rainfall and stream discharge (Marsh 1991). Sediments entering the stream's channel originate most commonly from bank erosion, bank failures, and basin runoff (May 1996). As previously mentioned, higher levels of sediment loads are associated with the initial development phase of urbanization. This may be followed by a net reduction of sediments, however, as permanent impervious surfaces are developed and act to prevent basin sediment erosion (Arnold et al. 1982). Higher proportions of impervious surfaces, however, may increase erosion rates due to channel incision and bank failure from alterations to the hydrologic regime, and to increases in the competence of flow, which eventually result in channel armoring and/or constraint (May 1996). Channel armoring can simply be naturally occurring bank reinforcement with boulders or the more intensive installation of man-made channels constructed from concrete.

An increase in the level of erosion caused by urbanization is associated with an increase in channel size. Fox (1976) found that erosion rates for urban streams in Maryland can increase by a factor of 15 over similar non-urban streams, and that the channels of urban streams increased in channel dimensions three times faster. McGriff

(1972), in a review of five studies investigating the level of erosion and urbanization, noted that increases in sediment load within streams of heavily urbanized catchments could be anywhere from 10-100 times those of forested catchments.

It is important to emphasize the role of the riparian corridor in determining rates of sediment supply to stream channels. Inputs of coarse woody debris and organic matter function as retentive structures and can also lower the effective stream gradient (Gregory & Davis 1991). These retentive structures reduce stream power by both lowering the stream gradient and increasing channel roughness, thus reducing the stream's erosion potential (May 1996). In addition, roots that extend into the streambank reinforce the channel against erosive forces. Whipple et al. (1981) compared 25 reaches of streams in New Jersey, based on ratings of the level of erosion (from high to low), and found that high levels were correlated with urbanization and low levels were correlated with the presence of wide riparian buffers. May (1996) found an inverse relationship between the level of urban development and quantity (i.e. width) and the quality (i.e. age and structure of vegetation) of the riparian corridor.

The physical character of stream channels that drain urbanized catchments is a result of the interaction between stream power and sediment source. Streams, such as those in the early phases of the urbanization process that receive sediment loads that overwhelm stream power, will be characterized by accumulating sediments and an overall reduction of the dominant particle size of the streambed. Such channels will also show accumulations of fine sediments causing an increase in substrate embeddedness and the filling in of habitat present within the stream substrata (May et al. 1997). Streams, such as those in the post-development phase of urbanization, often do not have sediment

inputs adequate to dissipate the power of the stream, due to increases in PTIA which decreases the amount of sediment supply and increases the frequency and volume of storm flows. Such streams will have streambeds characterized by an increase in the dominant substrate size and will be devoid of sediment accumulation (Robinson 1976, Arnold et al. 1982). These streams will be predisposed towards increased channel erosion and widening unless, as found in many urban settings, the channel is reinforced by some type of channel support or armoring.

Influences on Habitat Quality

In the northeastern U.S.A., good quality or unimpaired stream habitat in high gradient streams is characterized by >50% cobble substrate and a high number of riffles (pool to riffle ratio between = 5-7). These cobbles must be relatively unembedded (<5% of substrate covered by fine particles; Barbour & Stribling 1994). Under such conditions, stream flow depth and velocity will be variable and will support a heterogeneous mix of habitats, such as pools and glides with slower flows, in addition to riffles with faster flow velocities (Barbour & Stribling 1994). Unimpaired conditions are associated with wide, uninterrupted riparian corridors (>18m) with vegetation covering the majority of the stream bank areas (>90%), a lack of channelization and evidence of overflows on the upper banks, stable bank material preventing mass failures, and the absence of large erosional/depositional areas covering the stream bottom (<5%; Barbour & Stribling 1994). This habitat template, largely controlled by sediment supply and the hydrologic regime, has direct influences on biotic community structure and ecosystem function (see

Influence on Benthic Macroinvertebrates). Alterations to this habitat template will have large influences on community structure (Poff et al. 1997)

Protocols have been developed that evaluate stream habitat quality (Platts et al. 1983, Plafkin et al. 1989, Barbour & Stribling 1994, US EPA 1997). These protocols are “multimetric” and involve rating different parameters from good (similar to reference conditions) to bad (highly divergent from reference conditions) on a weighted, numerical scale and then combining scores to produce an overall rating. Parameters often include the frequency of riffles, riparian width, sediment coverage, embeddedness of substrate, and bank stability. Habitats strongly affected by urban land-use tend to be relatively homogeneous in terms of habitat, supporting predominately glide habitats with relatively constant flow velocities (Barbour & Stribling 1994). In addition to habitat simplification, riparian clearing, large areas of sediment cover, high levels of substrate embeddedness, and mass failures in bank material are often apparent (Barbour & Stribling 1994).

In an evaluation of 120 reaches in 22 streams in the Pacific Northwest, May et al. (1997) found a direct relationship between increases in PTIA and decreases in habitat quality, as indicated by qualitative comparisons to reference habitat conditions. They also found that streams with higher than expected habitat scores usually had riparian vegetation buffers. Streams with a continuous source of coarse woody debris and roots maintain adequate riffle-pool structure. Schueler (1994) similarly noted a reduction of habitat quality as PTIA approached 10-15%. Steedman (1988) failed to find a single habitat score above “poor” when catchments had PTIA exceeding 35%. These studies consistently found that habitat quality became reduced as riparian corridors were urbanized. The most significant impact on habitat seems to be a reduction in the number

of riffles and an increase in the number of pools due to higher rates of sedimentation (Pitt 1995).

Urbanization and Water Quality

Urbanization is associated with the degradation of the quality of surface waters. Although the Clean Water Act (PL 92-500) of 1972 has resulted in a significant reduction of the harmful substances reaching streams as point source pollution, it has been less effective in controlling non-point sources (NPS) (Rosebloom 1982). By definition, NPS pollution arises from diffuse sources and often reaches the stream as runoff (Marsh 1991). Common effects include increases in the following: oxygen demand, nutrient loading, concentrations of organic compounds and sediments, and the amount and variety of heavy metals reaching streams (Marsh 1991). Urban runoff was listed as the most significant polluter of New England rivers, affecting nearly twice the number of river miles as industrial point source discharges (US EPA 1998). Nearly 30% of all water quality impairments in the U.S.A. are linked to urban stormwater discharges (US EPA 1992).

The effects of exposure to NPS pollutants can be manifested as acute toxicity, as in the case of massive fish kills, or as the cumulative effects of long-term chronic exposure. Pitt et al. (1995) noted that stream biota are influenced more often by long-term chronic exposure to NPS pollution rather than conditions of short-term acute toxicity. NPS pollution and stormwater runoff have received much attention, with most emphasis being on sources, concentrations, constituents, fate, and dynamics. An

exhaustive treatment of this subject is beyond the scope of this review. The information that follows, however, emphasizes the cumulative influence of runoff on aquatic systems.

Spatial and Temporal Variability in Water Quality

The effect of stormwater runoff on stream water quality has significant spatial and temporal components. In a comprehensive review of the characteristics of NPS pollution and stormwater runoff, Porcella & Sorenson (1980) noted that factors influencing the nature and extent of runoff could be divided into watershed, hydrological, and transport factors.

Watershed factors include PTIA, traffic and road density, industrial activity, and other miscellaneous factors such as street sweeping (Porcella & Sorenson 1980). Schueler (1987) noted a proportional relationship between the extent of imperviousness within a catchment and the level of water quality degradation. In addition, several studies have found correlations between the type of urban land-use within the catchment and the level of pollution originating from runoff (Griffin et al. 1980, Bannerman et al. 1993, Wimberely & Coleman 1993, Claytor 1996). In Wisconsin, for example, runoff from an industrial roof contained $1348 \mu\text{g L}^{-1}$ of zinc as compared to runoff originating from a residential roof, which contained only $363 \mu\text{g L}^{-1}$ from a storm of similar intensity (Claytor 1996).

Hydrologic characteristics include storm intensity and storm frequency (Porcella & Sorenson 1980). These characteristics are unaffected by the influences of urbanization. Storm frequency is an important factor, however, because the accumulation

of pollutants on impervious surfaces can often be proportional to the time from the last storm (Roesner 1982).

In addition to watershed and hydrologic factors, those affecting the transport of stormwater are also important determinants of stream water quality. Porcella & Sorenson (1980) noted that factors such as the level of drainage efficiency, the use of detention ponds and other best-management-practices for runoff management, and the presence of combined sewer overflows (CSOs) could influence the amount of degradation from NPS pollution.

CSOs, commonly used for runoff management, combine storm sewers with raw sewage collection pipes. During small flows, the runoff reaching sewers is sent to the treatment facility along with the municipal sewage. During large flows, however, the runoff volume may exceed the capacity of the sewer and the runoff as well as the raw sewage is discharged into the stream through an overflow device. Such discharge is considered a significant threat to surface water (Ellis & Marsalek 1996). Hvitved-Jacobsen et al. (1994) showed that runoff from a CSO event contained up to two times the TSS concentration, five times the concentration of nitrogen (N) and phosphorous (P) of average stormwater, and similar concentrations of metals common to urban stormwater (Pb, Zn, Cd, and Cu). Typically, recovery periods from CSO discharges of both sewage and stormwater runoff take 5-7 days due to increased pollutant concentrations and higher flows associated with CSO overflow events, compared with recovery periods of only 1-2 days following stormwater runoff alone (Ellis & Hvitved-Jacobsen 1996).

For certain constituents of urban runoff, there is a "first flush" phenomenon characterized by decreasing concentrations with time (Porcella & Sorenson 1980,

Schueler 1994, Herricks 1995). Schueler (1994), in a technical note regarding research on NPS pollution in Texas, observed that this phenomenon is more prevalent when PTIA exceeds 50% and only for select constituents of runoff. Griffin et al. (1980) concluded that an important factor regarding the first flush effect is the phase (solid or dissolved) of the pollutant, with the level of solids being proportional to flow but levels of soluble pollutants being more dependent on factors affecting solubility.

Dissolved Oxygen (DO)

Keefer et al. (1979) investigated the relationship between DO concentration and urbanization for streams in the midwestern and northeastern U.S.A. They found that 60% of the study streams had greater than average DO deficiencies and that most had DO levels below 75% saturation. In addition, DO levels of $<5 \text{ mg L}^{-1}$ were common. This is particularly significant because five mg DO L^{-1} is often identified as the threshold level of DO required for a healthy biological community, with two mg DO L^{-1} being the critical value for maintaining aerobic life in streams (Roesner 1982).

Depletion of DO can be caused by several mechanisms including inputs of urban runoff. Keefer et al. (1979) noted a correlation between low levels of DO and inputs of stormwater runoff. Heaney (1980), in his summation of results from the US EPA National Urban Runoff Program including 28 cities nationwide, similarly found that the lowest levels of DO often occurred after storms. Ketchum (1978), however, in a study of urban streams in Indiana, was unable to link difference in DO levels with runoff. In spite of these conflicting reports, urban runoff has been noted to increase the biological oxygen demand (BOD) of streamwater. The BOD of runoff is an indication of the level of

organic enrichment reaching the stream (Porcella & Sorenson 1980). The BOD of streamwater, after a five-day incubation period (BOD_5), for the urban reaches of Coyote Creek in California was five times that of the non-urban reaches (Pitt 1994). In their review, Porcella & Sorenson (1980) presented summaries of studies to show that BOD_5 associated with urban streams could be 10 times higher than non-urban streams (Porcella & Sorenson 1980).

Depletion of DO is generally associated with increased levels of organic pollutants entering the stream through runoff, but can also be caused by organic pollutants residing in the sediments from previous runoff events. Alteration of the scour and fill dynamics of the stream, and additions of sediment due to erosion, can yield pulses of suspended sediment. This resuspended bed material may release organic constituents that may increase BOD (Roesner 1982).

Stream algae can also cause DO depletion as a function of urban influence. DO levels exhibit a diurnal cycle as a function of the release and uptake of oxygen due to plant respiration and photosynthesis (Allan 1995). With the combination of increased sunlight reaching the stream from riparian clearing and increases of nutrients associated with urban runoff, algae can reach a level that produces oxygen demand at night that can substantially reduce the level of DO within the stream (Klein 1979).

Influences on TSS

Fine suspended sediments (often defined as particles $< 0.85\text{mm}$ in diameter) were noted by May (1996) as the most significant influence of land-use on water quality. Wolman & Schick (1967) found the level of TSS to be a function of catchment land-use.

TSS levels are also influenced by channel erosion. Porcella & Sorenson (1980) noted in their review that most TSS levels associated with urban land-use are between 2.5×10^3 and $3.6 \times 10^4 \text{ mg L}^{-1}$. Concentrations of TSS are considered low at $<10 \text{ mg L}^{-1}$ and high at $>100 \text{ mg L}^{-1}$. Solo-Gabriele & Perkins (1997) noted that the intensity of storms and the volume of runoff can control TSS dynamics because higher volumes of discharge or runoff have the power to mobilize and entrain sediments.

Increased TSS may cause decreases in primary productivity due to reduced transparency of the water column, and decreases in habitat availability due to the filling of interstitial spaces in cobble riffles (Herricks 1995). In addition, ionic forms of heavy metals may have an affinity for charged clay-particles that make up much of TSS. As such, elevated levels of TSS can exacerbate contamination levels (Wilber 1980). TSS can settle out from the water column during weaker flows and can be resuspended after storms. Potentially, these resuspended TSS loads with their bound contaminants can produce harmful "slugs" to downstream communities (Wilber 1980).

Nutrient Loads from Urban Development

The US EPA has found eutrophication of water bodies to be the most widespread form of water quality degradation (US EPA 1998). Nutrients such as N and P as NPS pollution are the primary cause of eutrophication (Carpenter et al. 1998). Dissolved N is most commonly in the form of $\text{NO}_3\text{-N}$, and can also be found as $\text{NO}_2\text{-N}$, NH_4 , and organic N. Dissolved P is usually in the form of soluble reactive P (SRP) or orthophosphate and occurs in New England often in concentrations much lower than dissolved N. Marsh (1991) noted that annual inputs of N and P from urban land-use

could reach levels as high as 29.9 kg hectare⁻¹ yr⁻¹ and 3.0 kg hectare⁻¹ year⁻¹, respectively. Pitt (1994) described a significant increase in nutrient concentrations in Coyote Creek in California associated with urban runoff, with far greater inputs of NO₃-N than NO₂-N and NH₄ inputs. There was also an increase in the total P concentration associated with runoff inputs. In the comparison of urban Kelsey Creek to non-urban Bear Creek in Washington, Sloane-Richey et al. (1981) noted a significant increase in the NO₃-N and NO₂-N concentrations in the urban stream, as well as a three-fold increase in SRP. When comparing catchments with forest, agriculture, and urban land-uses in North Carolina, Lenat & Crawford (1994) found lower concentrations of total N in the urban compared with the forested stream. However, the urban stream contained a greater proportion of the N in dissolved forms (NO₃-N, NO₂-N, NH₄). The concentrations of both total N and P in the urban stream were, however, not as high as in the agricultural stream (Lenat & Crawford 1994). The extent of nutrient inputs is thus highly dependent on the nature of the land-use within the catchment (Kibler 1982, Marsh 1991).

High levels of nutrients in surface waters can alter the rate of primary production and produce algal blooms that deplete DO and produce large-scale fish kills (Carpenter et al. 1998). Degradation to this extent is rarely seen in small streams. However, the loading of nutrients within streams plays a role in the fate of the ultimate receiving water (Klein 1979). As mentioned previously, increased primary productivity can have an effect on the diurnal patterns of small streams DO. In addition, nutrient levels can alter the energy dynamics within the stream and benthic community and change trophic structure and complexity (Herricks 1995).

Toxics Concentration and Loading

Along with DO, TSS, and nutrients, streams affected by stormwater runoff often have elevated levels of toxic substances. Especially important in this regard are heavy metals. Heavy metals can be found in both the sediments and water column of streams (Porcella & Sorenson 1980). Sediment metal concentrations have been found to be inversely proportional to sediment size (Wilber 1980). Common metals associated with urban land-use include Pb, Zn, Cu, Ni, Cr, Cd, Mg, and Fe (Pitt et al. 1995). Wilber (1980) noted increases in these metals of 2.8 to 6.7 $\mu\text{g L}^{-1}$ below urban areas along the Saddle River in New Jersey. The largest increases in concentration were noted for Pb, the lowest being for Fe and Mg. Both of these latter metals are often associated with natural substrata (Wilber et al. 1980). Pitt & Bozeman (1982) noted much higher concentrations of Pb, Zn, Cu, Cd, Fe, Ni, and Hg following wet weather in their study of Coyote Creek in California. Garie & MacIntosh (1986) noted elevated heavy metals downstream of an urban area in New Jersey, as measured by scraping artificial substrate. Casper (1994) found significantly higher concentrations of Mg, Pb, and Zn, in his comparison of two bays within the same lake receiving runoff from both urban and forested land-use types. Lenat & Crawford (1994) found slightly higher levels of Cr, Cu, and Pb in the urban as compared to a rural catchment. They also noted higher levels of dissolved metals during wet weather, indicating runoff as the source of the metal contamination.

As with other forms of NPS pollution, the types and concentrations of heavy metals are highly dependent on the nature of the land-use within the catchment. Line et al. (1996) evaluated the runoff associated with five different industrial land-uses in North

Carolina and found variation in the type and concentration of pollutant loads. Metal contamination of streams often results in long-term chronic exposure of the benthic community and can lead to the bioaccumulation of metals within benthic organisms, the reduction of sensitive species within the community, and a large-scale reduction of the diversity of the benthic community (Klein 1979, Herricks 1995).

Influences on Temperature

Urbanization may alter the thermal regime of streams. Pluhowski (1968), in his comparison of five Long Island streams with varying levels of urban land-use, showed that summer temperatures may increase 10-15 °C and winter temperatures may decrease by 3-5 °C in urbanized streams compared with reference streams. These differences were attributed to the creation of ponds, clearcutting of riparian vegetation, increases in the volume of urban runoff, and reductions in groundwater recharge (Pluhowski 1968). Klein (1979) noted a difference of 11°C between a well-shaded section of a stream and an open pasture setting. As previously noted, the connection between riparian removal and urban development has been well established (May 1996). The opening of the canopy can cause significant increases in water temperature. In addition to sunlight reaching the stream directly through open canopies, temperatures in streams can be elevated by stormwater runoff. Large areas of cement and pavement can act as heat sinks and warm the runoff as it travels across the land surface. Herricks (1995) identified thermal pollution from NPS pollution as a possible risk and noted that it can influence rates of biochemical processes and result in the loss of sensitive benthic taxa.

Influences on Benthic Macroinvertebrates

Benthic macroinvertebrates are particularly useful as integrative indicators of the effects of a wide range of stresses on aquatic systems. As such, they have often been employed in studies of the influence of urbanization. Macroinvertebrates tend to have relatively limited geographic ranges, are abundant and easily sampled, and are easily identified with valid and reliable taxonomic keys (Platt et al. 1983). They also have life spans sufficient in length to provide a record of relatively recent events influencing the stream (Pratt & Coler 1976). In addition, much effort has been expended on determining the response of specific taxa to many different factors (Rosenberg & Resh 1993).

Attributes of the benthic community such as diversity, richness, and taxa abundance are often used to evaluate the extent of anthropogenic influence. Diversity is most simply the taxonomic richness, or number of different taxa, within the community (Plafkin et al. 1989). Numerical representations of community diversity can be calculated several ways (e.g. Shannon-Weaver Index of Diversity), and often integrate community richness and evenness (Pratt & Coler 1976). Evenness is the relative frequency with which organisms are distributed among taxa (Pratt & Coler 1976).

In conjunction with the above community attributes, specialized biotic indices are also commonly used as combinations of measures (“metrics”), called “multi-metric” indices (Plafkin et al. 1989). The Helsinhoff Biotic Index is a metric often employed in evaluating the influence of urban pollution on benthic communities (Plafkin et al. 1989). It is calculated by multiplying the abundance of a given species by a coefficient describing the tolerance of that species to anthropogenic stresses. Examples of other common metrics include %dominant taxa and %EPT, a measure of the percentage of the

community that are Ephemeroptera, Plecoptera, or Trichoptera (Plafkin et al. 1989).

These taxa have been found to be relatively pollution-intolerant and achieve their greatest richness in non-stressed aquatic systems (Rosenberg & Wiens 1993). Higher proportions of oligochaetes, isopods, and selected Diptera (specifically selected Chironomidae genera) have been found to indicate pollution stressed communities (Rosenberg & Wiens 1993).

In addition to the above measures, the distribution and relative abundance of functional feeding groups can provide information regarding the nature and extent of a given stress. Cummins (1974) developed functional feeding group classes for macroinvertebrates depending on the type and quality of food utilized. Classes include predators, shredders, collector-gatherers, collector-filterers, and grazers (Cummins 1974). Changes in community structure can indicate shifts in food quantity and quality that can be difficult to discern otherwise.

Macroinvertebrate Community Alterations

There is overwhelming evidence that the cumulative influence of urbanization causes changes to the benthic macroinvertebrate communities. Klein (1979) was the first to report the relationship between changes in macroinvertebrate community and urbanization in his study of 27 streams in Maryland. He found a dramatic reduction in diversity that was correlated with urban intensity (Klein 1979). Jones & Clark (1987) investigated three watersheds in the Washington, DC area, with varying land-uses, and also found a reduction of richness and an overall reduction of the pollution intolerant EPT taxa within streams with more urbanized catchments. Hachmoller et al. (1991)

performed a longitudinal study of a stream in Washington state, with sites on a forested upstream reach, a cleared reach, and a downstream urban reach. They found a reduction of shredders and collectors in the reach with a reduction in the inputs of allochthonous material consistent with intensive riparian vegetation clearing, and a reduction of the EPT taxa within the urban reach. In addition, they noted an increase in the abundance of oligochaetes and gastropods within the urban reach. Magnun (1989) studied five catchments with varying land-use in Virginia and found a reduction in the abundance of Ephemeroptera and Plecoptera and an increase in Trichoptera and Chironomidae within the urban catchments. Magnun (1988) also found an overall drop in taxa richness from 20 to eight taxa when comparing a forested to an urbanized catchment. Kemp & Spotila (1997) also found a reduction in sensitive macroinvertebrate taxa, with an increase in oligochaetes and isopods in their study of Valley Creek in Pennsylvania. Benke et al. (1981), in a study of 21 watersheds in Georgia ranging from 0-98% urban land-use, noted a reduction in the total number of taxa at the species as well as the family level. Benke et al. (1981) found little correlation between water quality parameters and the level of urbanization but found a strong relationship between the condition of the macroinvertebrate community and urban intensity. Pitt & Bozeman (1982), in their longitudinal study of Coyote Creek in California, found a similar decrease in the abundance of Ephemeroptera and Plecoptera as well as a dramatic increase in oligochaetes. Oligochaetes comprised 97% of the total urban-influenced community (Pitt & Bozeman 1982). In addition, they noted a decrease in the taxonomic richness of urban sites, dropping from 15-30 in the non-urban to five or less in the sites influenced by higher levels of urban development (Pitt & Bozeman 1982).

Although the result of these studies are consistent, the exact mechanism of the influence on macroinvertebrate community structure has yet to be fully identified and may vary with study site. Habitat alteration and the addition of sediment, alteration of the food supply, and influences of NPS inputs are all possible causative agents.

Habitat Alteration and Sedimentation

Habitat alteration, as a result of urbanization, has been associated with changes in the macroinvertebrate community structure of urban streams. Maxted (1996), in a study of 38 streams in Delaware, found a strong correlation between the changes in habitat quality due to urbanization, as indicated by qualitative comparisons to reference habitat conditions, and the condition of the benthic community. Invertebrate richness decreased with decreasing habitat index scores (Maxted 1996). By investigating spatial patterns of macroinvertebrates among 11 streams in Illinois, Richards et al. (1993) showed a strong correlation between decreasing benthic community richness and decreases in substrate particle sizes that they associated with urban development. In a study of an Edmonton, Alberta stream, Whiting & Clifford (1983) found a decrease in pollution sensitive taxa as well as a decrease in diversity and richness. This was primarily due to an increase in oligochaetes and Chironomidae abundance which was linked to increased of organic pollution and silt deposition (Whiting & Clifford 1983). Finally, Smith & Kaster (1983), in a study of several Wisconsin streams receiving low, intermediate, and high-volumes of roadway runoff, found a loss of 50% of the pollution-intolerant taxa in the intermediate-volume stream due to sedimentation of the habitat. The high-volume stream, however, provided adequate habitat and there was little influence to the community structure and

richness. The biomass and abundance of invertebrates actually increased in this stream. The maintenance of adequate habitat quality was able to mask the effects of the runoff (Smith & Kaster 1983).

The effects of sedimentation on habitat have been linked to changes in the macroinvertebrate community. Lenat et al. (1981) found that an increase in sediment loads to two streams in Caldwell County, North Carolina had a variable influence. At high flows, a reduction in the density of benthic invertebrates was associated with the filling of interstitial spaces by sediments that reduced the amount of available habitat (Lenat et al. 1981). At low flows, depositional bars provided erosional habitat that resulted in an overall increase in density of invertebrates, as well as a shift in the community towards a greater percentage of grazers (Lenat et al. 1981). In a study involving the experimental addition of sediment to two channels in the Harris River of the Canadian Northwest Territories, Rosenberg et al. (1978) found that there was a significant increase in the amount of drift within the sediment-affected channel. In a study of the effect of sedimentation due to construction and logging in the Cullowhee Creek catchment in North Carolina, Lemly (1982) found a reduction in richness, diversity, and biomass that was consistent with previous studies. Like Lenat et al (1981), Lemly (1982) attributed this to filling of the interstitial spaces, as well as fine particle accumulation on the body and respiratory structures of macroinvertebrates.

Food Quality

Alterations to the macroinvertebrate community structure due to changes in in-stream food quality as a result of urbanization have been documented. In two streams in

Washington, Sloane-Richey et al. (1981) found an overall reduction in the quality of particulate organic matter (POM) with increasing urban development. This was manifested as a reduction in the organic carbon content and an increase in the fine sediment fraction (Sloane-Richey et al. 1981). The diversity and abundance of macroinvertebrates in these streams were highly variable but fluctuated in a manner consistent with fluctuation in food quality (Sloane-Richey et al. 1981). Within these same streams, Pedersen & Perkins (1986) found no influence from urbanization on the colonization of buried samplers by invertebrates, but there was a shift in the community structure towards organisms that were capable of using lower quality food sources, particularly oligochaetes and selected chironomids. Maltby et al. (1995) in their study of streams crossing beneath a heavily used motorway in England, noted a decrease in the litter processing efficiency of four of seven of the study streams with a concurrent shift in the community structure from shredders to collector-gatherers.

NPS as Agent of Degradation

Although the above studies imply some specific causal relationships between urbanization and stream macroinvertebrate communities, many studies point simply to the cumulative influence of NPS pollution. Pratt et al. (1981) concluded that the effects of urbanization on the benthic communities of the Green River in Massachusetts were primarily a function of the amount of runoff the sites received, with decreases in both the diversity of the benthic community and the abundances of sensitive invertebrate taxa being consistent with increases in runoff volume. Their finding that the community was characterized by a higher percentage of pollution tolerant taxa during low flow periods

supported this conclusion. Both Lenat & Crawford (1994), in their study of three catchments with varying land-use in North Carolina, and Garie & MacIntosh (1986), in their study of two catchments with varying levels of urban development in New Jersey, found reductions of diversity, the proportion of pollution intolerant species, and taxonomic richness with increasing urbanization. In both studies, however, physical and chemical attributes of the stream, measured at the time of sampling, were at levels insufficient to cause the observed influences to the invertebrate community. They concluded that changes to the community were the result of unmeasured NPS pollution (Garie & MacIntosh 1986, Lenat & Crawford 1994). Duda et al. (1982), in their comparison of an urban and a non-urban stream in Asheville, North Carolina, also linked a 75% reduction in taxonomic richness and a shift away from sensitive invertebrate species in the urban stream to inputs from leaky sewer system and NPS discharges.

Ultimately, urbanization influences the overall habitat and trophic resources, community structure, richness, diversity, and abundance of macroinvertebrates. Urbanization causes a shift from a diverse benthic community of pollution sensitive taxa and taxa requiring high quality food sources and habitat structure to a more homogeneous community of taxa considered to be pollution-tolerant. The tolerant taxa are often those able to cope with reduced food quality and sediment accumulation. Although the exact mechanism of how urbanization causes this shift in the benthic community has yet to be fully determined, this review provided consistent evidence of the nature of the alteration. Similar research has been done that points to similar disruptive influences of urbanization to stream fish communities (Scott et al. 1986, Steedman 1988, Limburg & Schmidt 1990, Imhof et al. 1991, Lucchetti & Frustenberg 1993, Weaver & Garman 1994).

Imperviousness as a Cumulative Environmental Indicator

Arnold & Gibbons (1996), Booth and Reinelt (1993), and Schueler (1994) all provided literature reviews that concluded that PTIA may serve as a predictor of aquatic system impairment from urban land-uses. When factors were used to standardize the extent of urbanization among studies, all of these authors found that 10-20% PTIA was associated with a wide array of biological, physical, and chemical impacts to stream systems such as channel stability, pollutant levels, and fish and invertebrate diversity.

Arnold & Gibbons (1996) pointed out that this relationship is not one of causality, but rather that PTIA can serve as an integrated measure of the cumulative factors associated with urbanization that affect aquatic resources. Even with the extensive research that has been performed on this topic, however, the exact mechanism of the influence of urbanization on stream systems has yet to be fully defined. This is most probably due to the inherent variability within stream systems and land-use characteristics.

Schueler (1992) proposed a "Scheme for Classifying Urban Stream Quality Potential," represented in Fig. 3, with 1-10% PTIA within the catchment indicative of stressed streams, 11-25% impacted streams, and 26-100% degraded streams. Although there is apparent agreement on the percentages of impervious area that are associated with various types of stream impairment, there has yet to be defined a conclusive threshold level of imperviousness that indicates a significant level of stream impairment. This is probably due to regionally specific characteristics, such as climate and geology, and the qualitative nature of how stream degradation is evaluated. Schueler (1994) called for more research to assess this potentially invaluable tool for aquatic resource

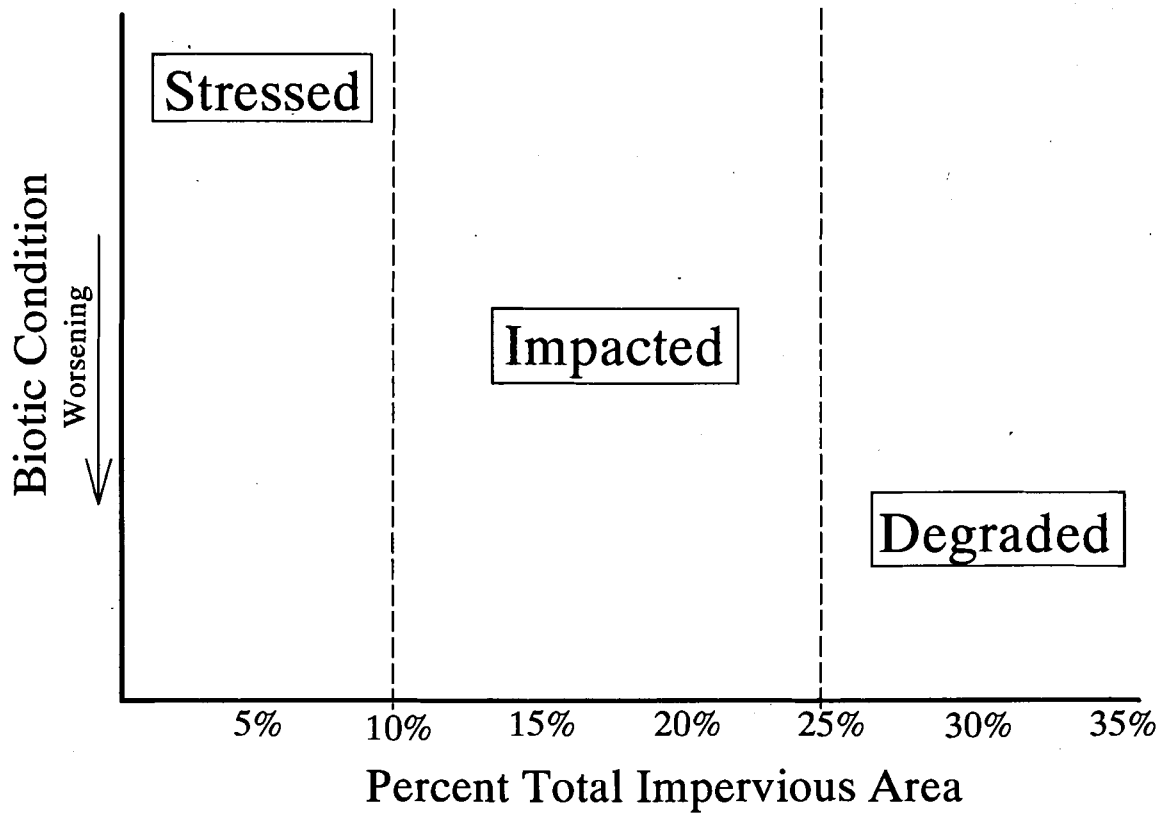


Figure 3. Theoretical relationship between the percent total impervious area (PTIA) of a catchment and the biotic condition of the stream. Modified from Schueler (1992). Dotted lines represent approximate thresholds separating each category of biotic condition.

conservation. He specifically targeted the northeastern U.S.A. as one of several areas for which further information is required.

In addition to the research needed within the northeast, there is currently a debate regarding the effectiveness of this indicator as predictor of stream impairment. Karr (1998) has encouraged caution in the use of PTIA as an environmental indicator, citing that it does not adequately predict invertebrate impairment at low levels. With these concerns in mind, further research into the usefulness of PTIA as a cumulative environmental indicator seems warranted.

Conclusion

Urbanization has been shown to drastically affect the condition of stream ecosystems through direct and indirect influences on the hydrological regime, habitat structure, and water quality. These effects collectively result in a change in the stream habitat template and the structure of the biotic community. Although much information has been gathered with regard to the result of the negative influences of urbanization on the benthic community, more research is needed to determine the exact mechanisms involved. The emergence of PTIA as a potential predictor of cumulative stress from urban land-use on stream systems is a powerful and exciting development. As urban areas continue to expand nationwide, PTIA as an indicator presents a powerful tool for resource managers to use towards stream conservation.

CHAPTER 2
THE RESPONSE OF FIRST AND SECOND ORDER STREAMS TO URBAN
LAND-USE IN MAINE, U.S.A.

Introduction

As previously stated, this study sought to answer two questions. First, is the level of degradation of the physical, chemical, and biological attributes of 1st and 2nd order streams in Maine related to increasing urban intensity across an urban gradient, as measured by the percent of the total impervious area (PTIA) within the catchment? Second, do adverse impacts of urbanization on Maine streams occur at a threshold of 10-20% PTIA as suggested by Schueler (1994)? The overall design involved a comparison of catchments under varying levels of urban development with reference catchments of similar ecological character but with minimal urban development. This study represents the first formal study in Maine of the influence of urban development on streams using PTIA as the indicator.

Methods

Catchment Selection

The goal of the catchment selection process was to select twenty catchments throughout southern and central Maine that represented varying levels of urban intensity from undeveloped to predominantly urban. Criteria for selection included region, stream width, depth, discharge, gradient, substrate, and habitat structure. Efforts were also made to select catchments with streams that were not influenced by obvious point sources of

pollution. Due to a paucity of urban areas within the state that could provide replicate study catchments within a single urban area, catchments were selected in regional blocks that included at least one reference catchment as well as catchments exhibiting low, moderate, and high levels of PTIA. Reference catchments are defined as catchments that are predominately forested with <5% PTIA (Schueler 1994). For the purpose of the initial catchment selection, urban intensity was estimated from 1:24,000 scale United States Geological Survey (USGS) topographic maps, *The Maine Atlas and Gazetteer* (DeLorme 1997), and field inspection.

Study Catchments

Twenty catchments of the over 300 assessed met the initial selection criteria. The estimated levels of urbanization among these catchments ranged from reference to heavily developed. Catchment blocks were selected in the vicinity of the cities of Bangor (area ~90 km², population ~33,000 residents), Anson/Madison (~24 km² and ~21,000 residents), Augusta (~143 km² and ~21,000 residents), and South Portland (~31 km² and ~23,000 residents) (Fig. 4a). The South Portland catchments drain into the Fore River and are located in the southern portion of Maine in the coastal plain. Augusta and Anson/Madison catchments are all located within the Kennebec River drainage. Augusta is in the coastal upland area whereas Anson/Madison is located farther west in close proximity to the western mountains of Maine. The Bangor catchments are located in the Penobscott River drainage in the coastal upland area. All blocks are located within the mixed wood plains ecological region as described by the Commission for Environmental

Cooperation (1994), although the Anson/Madison block is close to the border of the Atlantic highlands ecological region (Commission for Environmental Cooperation 1994).

Impervious Surfaces Analysis

Measurement of PTIA was conducted using 1:7,200 scale aerial photographs (Natural Resource Conservation Service, Maine). These photographs were taken from 1991 to 1998. The catchment boundaries were delineated using the watershed divide technique (Stanford 1996) onto 1:24,000 scale USGS topographic maps. Mylar plastic was placed over the appropriate aerial photograph and the catchment boundary transcribed. Large and obvious expanses of impervious surfaces within the catchment, such as parking lots and industrial buildings, were delineated and measured with a planimeter. Area of homogenous urban intensity and the total area of the catchment were also measured. The areas of homogenous urban intensity, including residential neighborhoods consisting of one, 1/2, 1/4, and 1/8 acre lots were multiplied by an empirically derived impervious surfaces factor. PTIA was calculated using the equation below:

$$PTIA = \frac{\Sigma (\text{area of large impervious areas}) + (\text{areas of homogenous density} \times \text{impervious factor})}{\text{catchment area}}$$

Although impervious surface factors have been reported from other regions (Soil Conservation Service 1975, City of Olympia 1994), they tend to be variable (Table 2).

Table 2. Estimates of percent impervious surfaces associated with common residential densities. TR-55 factors reported in Soil Conservation Service (1975) and HSPF model and Olympia Study factors reported in City of Olympia (1994).

Land-use	Percentages of Impervious Surface Area			
	this study	TR-55	HSPF Model ¹	Olympia Study ¹
Low-density (<1 unit/acre)	13.40%	20%	10%	n/a
Mid-density (4 units/acre)	28.20%	38%	40%	40%
High Density (8 units/acre)	32.20%	65%	60%	48%

¹ HSPF Model and the Olympia Report listed densities for mid-density to be 3-7 units/acre and high-density to be 8-30 units/acres.

Because of this, factors were calculated for this study by selecting small study areas of homogenous residential density and measuring impervious surfaces in the field.

Land-use Classification

A GIS was constructed that provided percentages of the study catchments that were covered by forest, agriculture, or wetland. This GIS was based on Erdas Imagine[®] and the Maine GAP vegetation classification (Hepinstall et al. 1999). The Maine GAP classification is a raster image of a 38-category vegetation classification of the state based upon Landsat TM satellite data (30 m pixel resolution) from 1991 and 1993 (Hepinstall et al. 1999). For this study, the 38 categories of land cover were composited into four inclusive categories (urban, forest, agriculture, and wetland) based on the descriptions provided by Hepinstall et al. (1999). The selected study catchments were digitized from 1:24,000 scale USGS topographic maps and their proportional land cover was classified. These data were used to assess the suitability of reference catchments and to identify possible confounding land-use factors (e.g. agricultural land-uses being the dominant land-use within the catchment).

In addition, aerial photointerpretation and field surveys offered an opportunity to qualitatively describe the nature of the urbanization. Urban land-use within the study region was categorized as: residential, including light (containing lots of 1 to ½ acre in size), moderate (containing ½ acre lots), and heavy (containing ¼ acre lots), commercial (containing facilities such as shopping centers), industrial (containing facilities such as rail yards and factories) and airports.

Physical/morphological Survey

For each study stream, a 100-meter sampling reach was located near the catchment's outlet. Study reaches were located at least 10 meters from any road crossing, culvert, or stream junction. Each study reach was divided into 10 meter sections by transects placed perpendicular to the stream flow (11 transects total). During August to October 1998, bank-full width and depth, wetted width and depth, bank erosion and angle, percent substrate, and riparian width and forest-type were measured at each transect. Additional reach-level measurements included gradient (using a hand-held Sunuto[®] clinometer) and habitat structure mapping. These were conducted as described in Meader et al. (1993), Platts et al. (1983), and May (1997) at least 48 hours following the last rainfall. Low-flow discharge was measured using the velocity-area method as described by Gore (1996) with a Globe[®] flowmeter.

Two qualitative indices were used to further describe the physical condition of the stream. The Qualitative Habitat Index (QHI), as described in Barbour & Stribling (1994), for riffle/run prevalent streams yields a composite score that integrates 10 different metrics indicating stream habitat quality. Each of these are evaluated qualitatively and scored from 0 to 20. Metrics include evaluations of substrate availability and condition, channel condition, extent of erosion/deposition, and riparian condition. Values for the QHI range from 0 to 240 (Barbour & Stribling 1994). QHI categorical ratings are "optimal" (180-240), "suboptimal" (120-179), "marginal" (60-119), and "poor" (0-59). The Stream Reach Inventory and Channel Stability Index (SRICSI), as described by Pfankuch (1975), integrates 15 metrics and evaluates the channel for instability and erosion/deposition using a weighted scale. Composite scores of the 15 metrics range

from a low (stable) score of 33 to a high (unstable) score of 162 (Pfankuch 1975).

Categorical rating for the SRICSI include “excellent” (<38), “good” (39-76), “fair” (77-114), and “poor” (>115).

A modified Wolman pebble count (Wolman 1954) was used to quantify the particle size distribution of the bed material. Ten particles were selected at evenly spaced intervals from 10 transects and their intermediate axes measured. The intermediate axes length refers to the axis length that is neither the longest nor the shortest measured from height, width, or length of the particle (Potyondy & Hardy 1994). These data (100 pebbles for each study reach for each catchment) were used to estimate the intermediate axis lengths of the median (D_{50}) and the fine fraction (defined for this study as the particle size of the D_{10} or tenth percentile particle).

Water Quality Sampling

Water temperature, pH, pre-dawn dissolved oxygen (DO), specific conductance, nutrients (nitrogen and phosphorous), and total suspended solids (TSS) were measured in the Summer (August to September) and Fall (November) of 1998 and Spring (April to May) of 1999. All seasonal data for the water quality attributes were averaged to provide an average annual value for each catchment. Temperature, pH, DO, and specific conductance were sampled at the upstream boundary of the study reach at least 48 hours after any precipitation using hand-held YSI[®] meters. TSS were determined using Advantec[®] GF75 filters with a nominal pore size of 0.7 μm . TSS were sampled using a minimum of 250 ml and a maximum of 1L of stream water taken from the upstream boundary of the study reach. Samples were vacuum filtered into an acid-washed glass

flask in the field. Filters were dried in a forced-air oven (150 °C) for 24 hours and then weighed. The dried/filtered weight of the filter minus the filter's initial weight equaled TSS. Filtrate from the TSS procedure for each stream and season was collected in acid washed bottles (250 ml) and returned on ice to the Maine Soil Testing Laboratory (Deering Hall, University of Maine, Orono, Maine 04473) where they were analyzed for NO₃-N and soluble reactive phosphorus (SRP) using an Alpkem[®] flow-injection analyzer with detection limits of 0.005 and 0.05 mg L⁻¹ respectively and total phosphorus (TP) using a Jarrell-Ash[®] TJA 975 spectrophotometer with a detection limit of 0.05 mg L⁻¹. The Maine Soil Testing Laboratory analyzes standards of known concentrations (certified by the National Institute of Standards and Testing) between every 10-15 field samples to assure the accuracy of their nutrient analyses.

Macroinvertebrate Sampling

Benthic macroinvertebrates were used as a biological indicator of the influence of urbanization (Rosenberg & Resh 1993). Benthic macroinvertebrates were sampled on two occasions: Fall (November to December) of 1998 and Spring (April to May) of 1999. Six samples were taken from randomly selected riffle habitats within the study reach using a Surber-type sampler with a 230 µm mesh net and a 0.090 m² frame. Riffle habitats were chosen for sampling as they harbor a relatively more diverse community of benthic macroinvertebrates compared with other common stream habitats (Brown & Brussock 1991, Hynes 1970, Logan & Brooker 1983).

Substrata were disturbed within the Surber sampler frame to a depth of ~4 cm. The material collected in the net was washed through a sieve bucket (500 µm mesh).

Large debris was removed and the remaining sieved material was preserved in 70% ethanol. In the laboratory, organisms from three of the six samples taken per stream per season were removed and identified to the lowest practical taxonomic level (usually genus) using Merritt & Cummins (1996), Thorp & Covich (1991), Wiggins (1977), and Stewart & Stark (1993). Exceptions included the Oligochaeta, Gastropoda, and Collembola, which were identified only to the class or order level, and the Simuliidae which were identified to the family level. In addition, larvae of the family Chironomidae were identified to tribe or subtribe (Tanypodinae, Diamesinae, Orthocladinae, Chironomini, Tanytarsini). Samples with an extraordinary amount of organic material or density of organisms were subsampled. However, no less than 100 organisms were identified from any sample.

Data for benthic macroinvertebrates were used to calculate taxon density, number of unique taxa (taxa present in only one sample), total richness (number of taxonomic divisions identified per sample), and EPT Index values. The EPT Index is the number of taxa within the Ephemeroptera, Plecoptera, and Trichoptera. The EPT Index is routinely used as a measure of the effect of anthropogenic stress on benthic macroinvertebrate communities (Barbour et al. 1999). Analyses were conducted separately for Fall and Spring to assess seasonal changes in macroinvertebrate community structure along the urban gradient.

Study streams were assigned to categories describing the extent of degradation using taxa richness (Plafkin et al. 1989), similar in a manner to the United States Environmental Protection Agency (US EPA) Rapid Bioassessment Protocol used to quantify the level of degradation of stream communities. Plafkin et al. (1989) suggests

that the health of a given stream can be rated based on comparisons with reference streams. Using the US EPA's thresholds, streams with >80% of the reference richness were considered "non-impacted", between 60-80% "slightly impacted", 40-60% "moderately impacted", and <40% "severely impacted".

Analyses

Dependent variables for analysis included physical channel attributes (channel morphology, QHI, SRICSI, size of D₅₀ and D₁₀ particles), water quality attributes (DO, specific conductance, TSS, NO₃-N, SRP, and TP), and attributes of the Fall and Spring benthic macroinvertebrate communities (density, total richness, and EPT Index values). These dependent variables were regressed against PTIA (independent variable). In order to meet the assumptions of the least-squares regression, abundance data were transformed using the ln(x+1) transformation. Regressions were considered significant when *p* values were < 0.05. All analyses were performed using the Systat[®] statistical software package.

Results

Impervious Surfaces Analysis

The urban gradient encompassed by the 20 study catchments (Table 3) ranged from a low PTIA of 1% in Josh to a high of 31% in AirI (Fig. 4b). AirI, representing the high end of the PTIA gradient covered by the selected study catchments, has an overall PTIA that is considerably lower than what is found in more urbanized parts of the U.S.A. (e.g. May et al. 1996). Although an extensive field survey was conducted during the catchment selection phase of this study throughout the more developed portions of the

Table 3. Environmental characteristics of the study catchments and streams. Abbreviations for estimated urban intensity include: H = high, M = moderate, L = low, and R = reference. Abbreviations for Land-use include INDUS = industrial, COMM = commercial, AIR = airport, L RES = light residential, M RES = moderate residential, H RES = heavy residential, and FOREST = forested.

Name	Abbrev.	Est. Urban Intensity	Land-use	Metro Area	Area (km ²)	Stream Order	Average Bankfull Width (m)	Average Bankfull Depth (cm)	Low-flow Discharge (L s ⁻¹)	Gradient (%)	Average Temp. (°C)	pH ¹
Goodrich Bk	Good	R	FOREST	Anson/Madison	4.8	2	4.9	29.8	1.6	2.9	9.1	7.31
Joshua Bk	Josh	R	FOREST	Anson/Madison	11.7	2	5.6	34.7	0.6	3.0	10.0	7.29
Hoyt Bk	Hoyt	R	FOREST	Augusta	5.9	2	6.3	34.3	103.9	3.5	8.7	7.48
Piper Bk	Piper	R	FOREST	Bangor	18.3	2	6.3	40.7	28.9	2.6	9.6	7.41
Pittsfield	Pitt	R	FOREST	Bangor	9.9	2	5.2	43.4	39.3	3.1	unavail	7.61
Pollard Strm	Poll	R	FOREST	Bangor	18.1	2	7.0	40.8	57.5	2.0	13.8	7.02
Minnow Bk	Minn	R	FOREST	S.Portland	3.6	1	5.6	30.0	3.5	0.9	10.2	7.46
Getchell Bk	Getch	L	L RES	Anson/Madison	14.6	1	7.9	42.0	36.4	3.0	10.2	7.12
Jones Bk	Jones	L	L RES	Anson/Madison	10.1	1	5.8	44.9	30.5	1.6	11.4	6.39
Whitney Strm	Whit	L	L RES	Augusta	3.8	2	5.1	24.2	3.4	3.6	11.8	7.73
Reed Bk	Reed	L	L RES	Bangor	5.6	1	9.7	38.5	30.0	4.0	8.8	7.99
Pushaw	Push	M	H RES	Bangor	2.0	1	5.9	29.0	14.0	0.1	8.9	7.84
Penjajawoc Strm	Penj	M	COMM, L RES	Bangor	16.0	2	4.8	28.7	12.1	1.5	9.6	7.58
Trout Bk	Trout	M	COMM, L RES	S.Portland	5.0	2	3.5	29.5	17.2	1.0	9.2	7.61
Long Ck	Long	M	COMM, AIR	S.Portland	5.7	2	3.8	32.4	18.8	0.6	10.8	7.59
Kennedy Strm	Kenn	H	INDUS	Augusta	2.1	1	5.0	32.9	8.2	2.1	12.6	7.85
Brewer	Brew	H	INDUS	Bangor	2.1	1	2.6	32.0	1.8	1.5	8.9	8.01
Airport II	AirII	H	M RES, AIR	Bangor	8.8	1	5.9	35.2	5.3	1.5	9.4	8.04
Airport I	AirI	H	COMM, AIR	Bangor	5.5	1	9.1	35.3	190.5	4.0	11.2	7.93
Barberry Ck	Barb	H	INDUS	S.Portland	2.7	1	7.9	25.7	16.5	3.0	10.0	7.58

¹pH measurement made directly from stream flow.

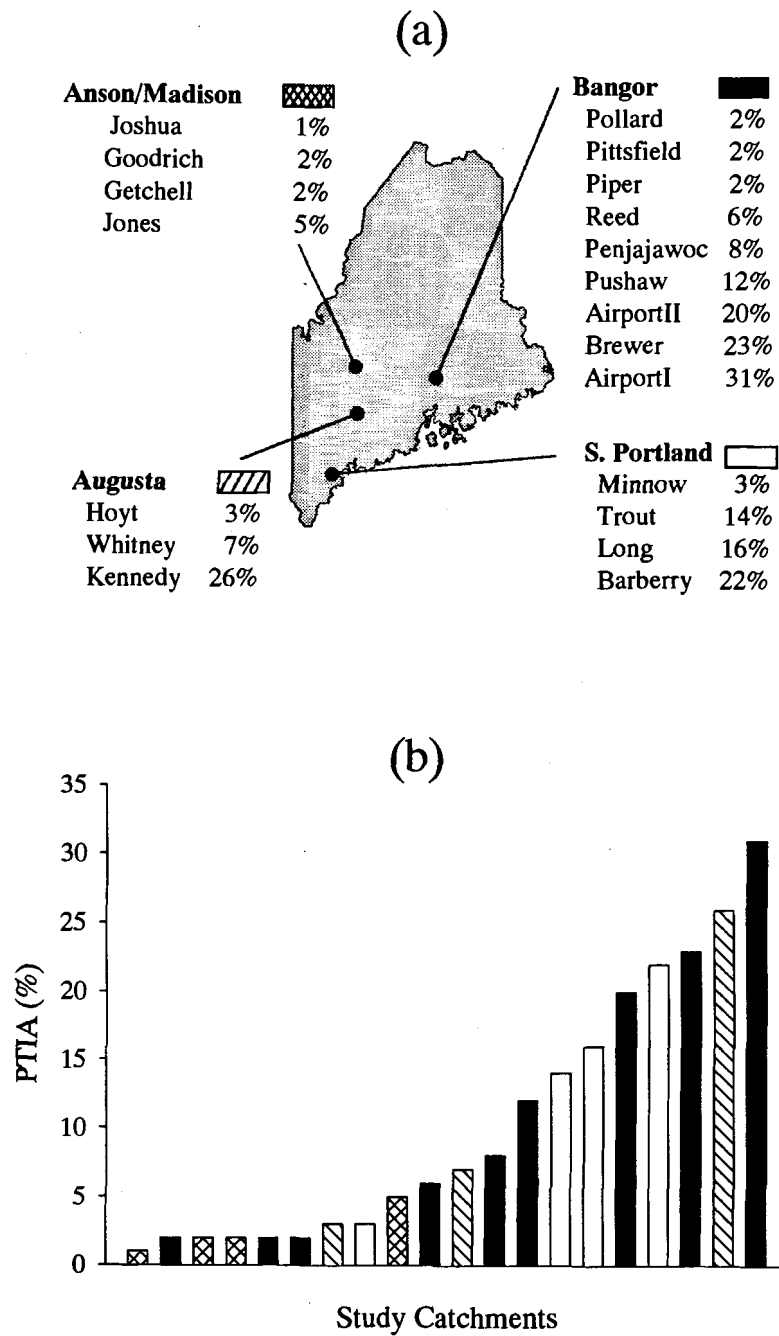


Figure 4. (a) Map of Maine showing locations of the study catchment blocks and PTIA values and (b) a graphical illustration of the urban gradient represented by the study catchments. Catchments are in order of increasing PTIA and color coded by regional location corresponding to the legend within (a).

state, AirI was determined to represent the upper limit of urban intensity in Maine. PTIA values were consistent with the urban intensities estimated during the initial selection of catchments, with the exception of Getch in the Anson/Madison area. This catchment was incorrectly classified as one with low urban intensity during the field survey but was determined to have PTIA <5% during the impervious surfaces analysis (Fig. 4a).

Land-use Classification

Land-use classification confirmed that eight catchments designated as reference (PTIA <5%) were predominately forested (Fig. 5 and 6b), with percentages ranging from 58% in Piper to 92% in Pollard. A significant relationship existed between the area of each catchment classified as urban by the Maine GAP classification and PTIA ($p = <0.0001$, $r^2 = 0.88$; Fig. 7). There were no trends between PTIA and the percentage of the catchments classified as agriculture or wetland (Fig. 6c-d). The area for Penj (PTIA = 8%), classified as urban land-use by the Maine Gap classification, was lower than would be expected based on trends in the data set as a whole (Fig. 7). It is assumed that urban land-use for Penj was underestimated by the Maine Gap classification. Similarly, the area of urban land-use for Trout was overestimated by the Maine Gap classification, most probably due to the inaccurate classification of large areas of agricultural land-use in the headwaters.

Long and Penj, catchments chosen for their moderate levels of urban intensity, were found also to have relatively high levels of agricultural land-use (Long with 28% urban and 44% agriculture and Penj with 12% urban and 25% agriculture; Fig. 6a, c). The risk of confounding effects from agricultural land-use in Long was considered

Note - This is a black and white copy of a color map. We do not have access to a color copy at our library - If you need a color reproduction, please email me -
 Betsy Beattie



Figure 5. Examples of study catchments with classified land-uses. Abbreviations found in Table 3. Stream channels are represented by white lines within the catchments. Catchments are ordered by increasing PTIA (Hoyt PTIA = 3%, Jones PTIA = 5%, Long PTIA = 16%, AirI PTIA = 31%).

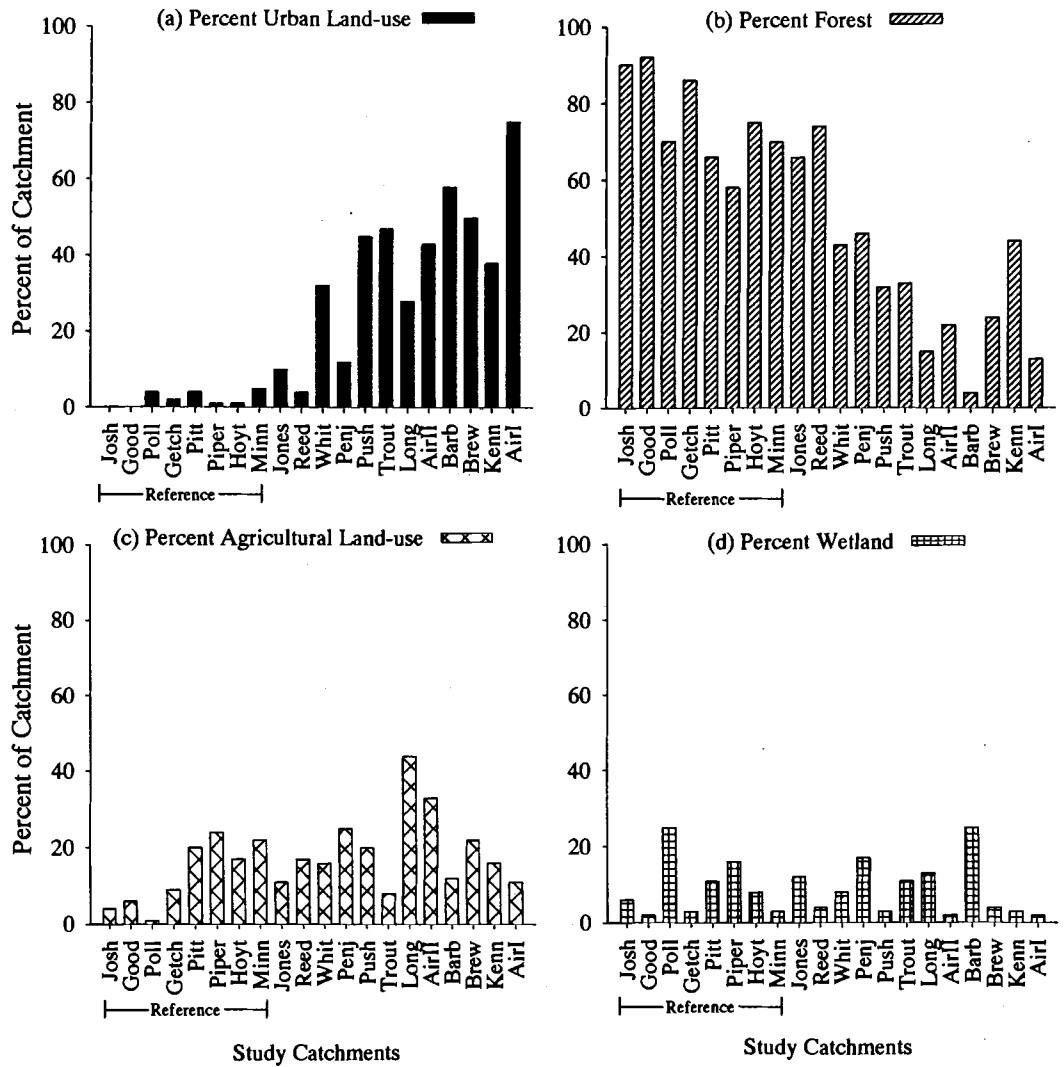


Figure 6. Percentage of catchments under various land-uses as determined by analyses of the Maine GAP vegetation classification. Categories include (a) percent urban land-use, (b) percent forest, (c) percent agricultural land-use, and (d) percent wetland. Abbreviations found in Table 3. Catchments are ordered by increasing PTIA values (see Fig. 4).

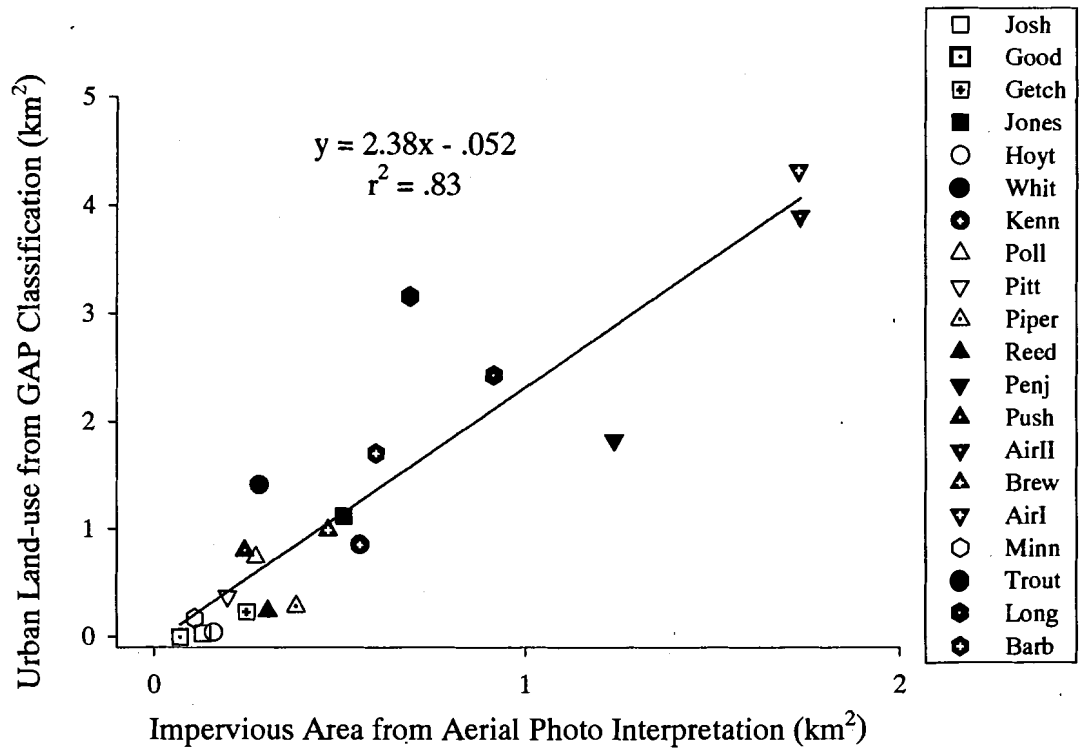


Figure 7. Relationship of the area of urban land-use of study catchments as determined by aerial photointerpretation to the area of urban land-use as determined by the analysis of the Maine GAP vegetation classification. Symbols are coded by shape to represent catchments within the same urban areas; squares for Anson/Madison, circles for Augusta, triangles for Bangor, and hexagons for South Portland. Open symbols reflect reference catchments (PTIA <5%). Abbreviations are found in Table 3. The solid line represents a significant linear regression ($p = 0.0001$) with the equation presented within the figure.

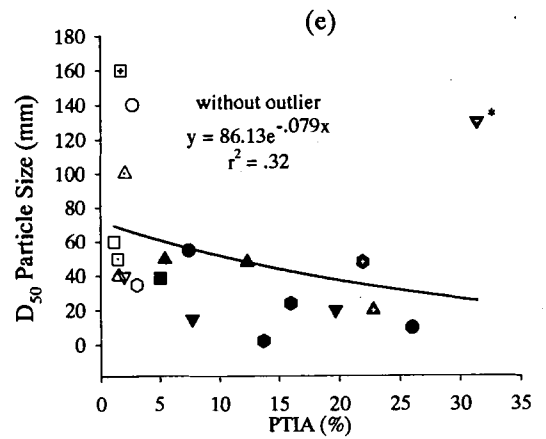
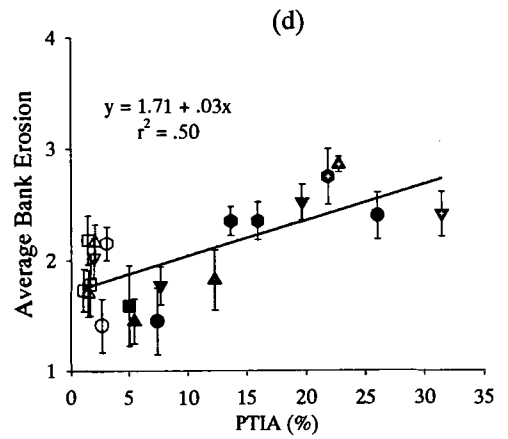
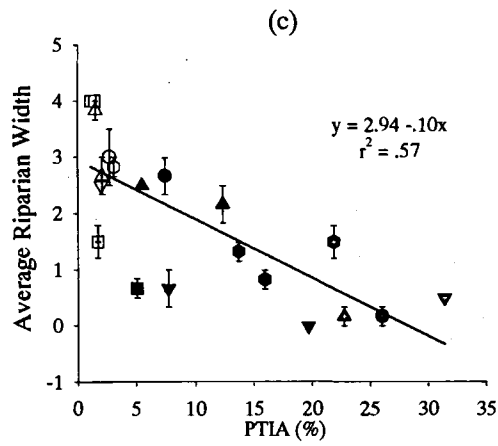
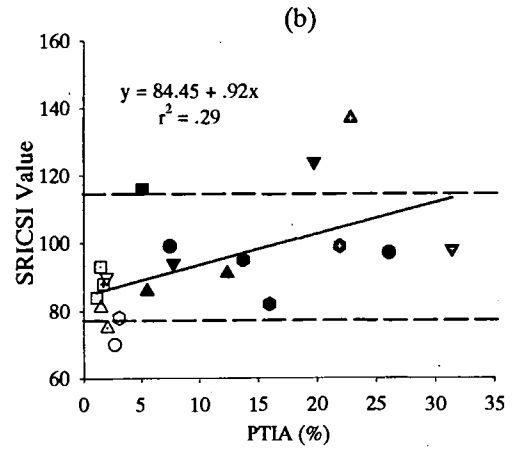
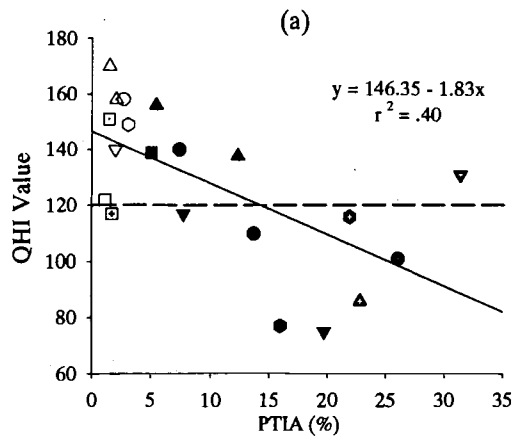
minimal, however, due to the extent and proximity of the urban land-use to the downstream study reach (Fig. 5c).

Physical/morphological Survey

Qualitative Habitat Index (QHI) values were negatively related to PTIA ($p = 0.003$, $r^2 = 0.40$; Fig. 8a). Compared to optimal QHI values of 180-240, all streams were within the “suboptimal” (QHI = 120-180 range) and “marginal” (QHI = 60-120) categories. Notable outliers include AirI, with a QHI value much greater (QHI = 131) than would be predicted by the trend in the data, and Long, AirII, and Brew which had QHI values that were lower than would be expected (QHI = 77, 86, 75, respectively). The relationship of QHI values to PTIA was supported by significant relationships between average riparian width ($p = <0.0001$, $r^2 = 0.57$; Fig. 8c) and average bank erosion ($p = <0.0001$, $r^2 = 0.50$; Fig. 8d) and PTIA. As PTIA increases, the width of the riparian corridor decreases from the categorical class of 4 (width = >100 m) to 0 (width = <10 m) and the level of bank erosion increases from the categorical class of 1 (low) to 3 (high) (Fig. 8c-d). Other stream habitat attributes, including indicators of habitat simplification (e.g. pool frequency, riffle to pool ratios, and riffle frequency) and channel dimensions (e.g. width to depth ratios), were not significantly related to PTIA.

Channel Stability Index (SRICSI) values and PTIA were positively related ($p = <0.0001$, $r^2 = 0.29$), suggesting a decrease in streambed stability with increasing urban intensity (Fig. 8b). SRICSI values ranged from “good” (SRICSI = 39-76) for study reaches in reference catchments to “fair” (SRICSI = 77-114) for study streams further along the urban gradient. Notable outliers include Jones, AirII, and Brew which were

Figure 8. Relationship of physical attributes of the study reaches to the study catchment PTIA values; including (a) Qualitative Habitat Index values, (b) Channel Stability Index values, (c) average riparian widths (4 observations/reach), (d) average bank erosion intensities (22 samples/reach), and (e) median particle sizes. Symbols correspond to legend in Fig. 7. Error bars indicate ± 1 S.E. Dashed lines in (a) and (b) correspond to qualitative classifications based on scores (see text). The categorical scale for (c) corresponds to 0 = <10m, 1 = 10-30m, 2 = 30-50m, 3 = 50-100m, 4 = >100m and for (d) 0 = no erosion evident, 1 = low level of erosion, 2 = medium, and 3 = high. Solid lines represent significant regressions (see text) with equations presented within the figure. Asterisks in (c) indicates a data point considered to be an outlier and removed from statistical analysis (see text).



categorized as “poor” (SRICSI = 115 - 162). These reaches had a greater level of instability than would be predicted by PTIA.

The Wolman pebble count indicated an inverse exponential relationship ($p = 0.0001$, $r^2 = 0.32$) between the median particle size (D_{50}) and PTIA (Fig. 8e). Because the stream channel of AirI was composed primarily of bedrock that prevented a meaningful measure of D_{50} , that stream was removed for statistical analysis of the pebble count data. No significant relationship existed between the sizes of the smaller D_{10} particles and PTIA.

Water Quality

Average specific conductance ranged from $59.9 \mu\text{S cm}^{-1}$ in Poll (PTIA = 1%) to $563.4 \mu\text{S cm}^{-1}$ in AirI (PTIA = 31%). Brew had an average specific conductance an order of magnitude greater than would be predicted from the trend within the data (Fig. 9a). The elevated level of specific conductance in Brew could not be attributed to any specific source, although the study reach is in close proximity to an interstate off-ramp. With this outlier removed, there was a significant relationship between PTIA and specific conductance ($p = <0.0001$, $r^2 = 0.74$).

Average pre-dawn DO ranged from $\sim 11 \text{ mg L}^{-1}$ in reference catchments to 5.8 mg L^{-1} in AirI. Average DO was negatively related to PTIA ($p = 0.001$, $r^2 = 0.48$; Fig. 9b). Average DO in Barb appeared to be lower than would be predicted, possibly due to the effect of a 200 m culvert located 10 m upstream of the sampling reach.

TSS ranged from $\sim 2.0 \text{ mg L}^{-1}$ in reference catchments to 5.0 mg L^{-1} in Barb (Fig. 9c). Providing that the single outlier Long was removed for statistical analysis, average

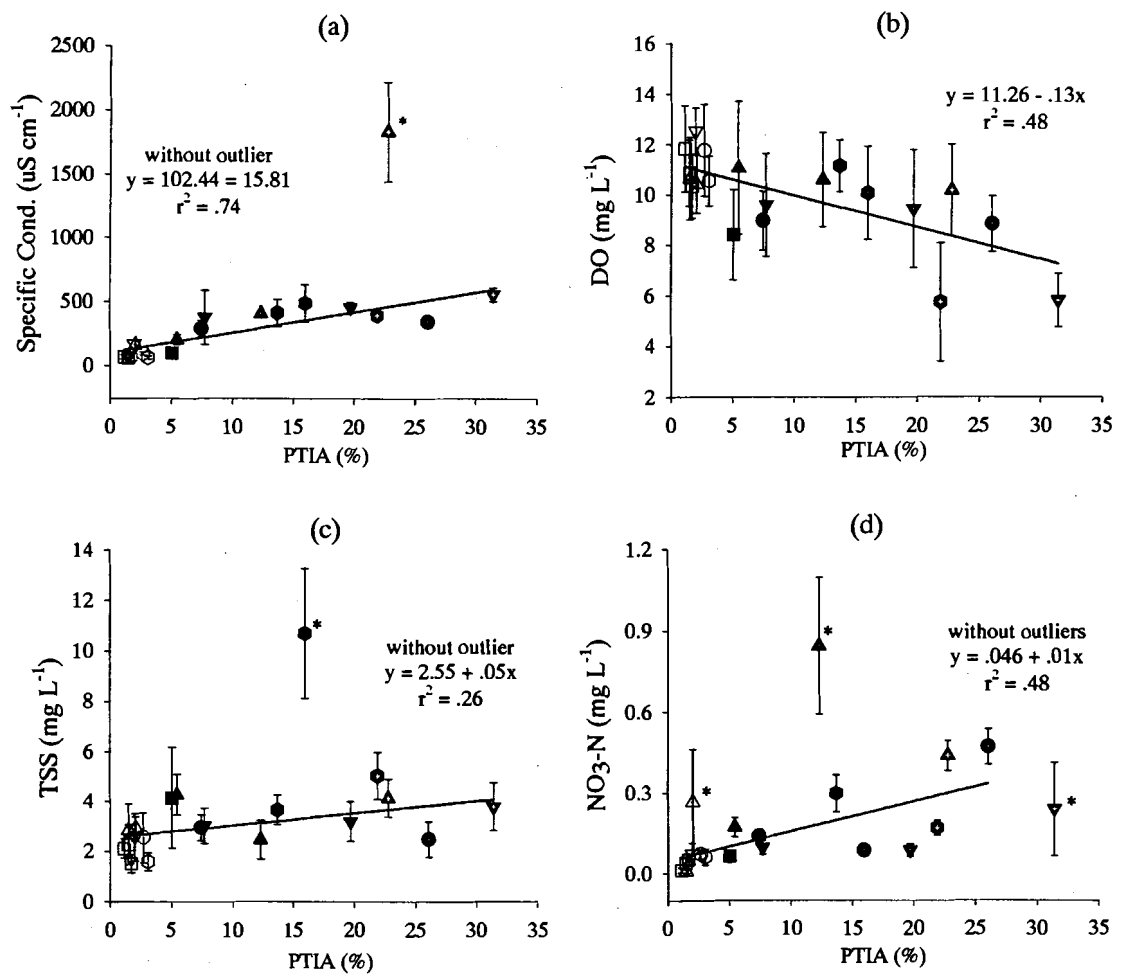


Figure 9. Relationship of water quality attributes to the study catchment PTIA values; including (a) average specific conductance (3 measurements/reach), (b) average dissolved oxygen (3 measurements/reach), (c) average total suspended solids (9 samples/reach), and (d) average NO₃-N (3 samples/reach). Symbols correspond to the legend in Fig. 7. Error bars indicate ± 1 S.E. Solid lines indicate significant regressions (see text) with equations presented within the figure. Asterisks indicate data points considered as outliers and removed from statistical analyses (see text).

TSS was positively but weakly associated with PTIA ($p = 0.025$, $r^2 = 0.26$). A large highway construction project had just begun within the catchment of Long at the time of sampling. TSS levels have been shown to be temporally variable with much higher levels present during the construction phase of urbanization (Wolman & Schick 1967).

$\text{NO}_3\text{-N}$ was detectable in all streams (detection limit = 0.05 mg L^{-1}). Apparent outliers included Piper, Push, and AirI (Fig. 9d) that were removed from statistical analysis due to their unexplained and considerably higher standard error (S.E.) values (Piper = 0.20 S.E., Push = 0.25 S.E., and AirI = 0.18 S.E.). These streams contained S.E. values associated with $\text{NO}_3\text{-N}$ measurements that were greater than three times that of the average S.E. for the $\text{NO}_3\text{-N}$ data (average S.E. = 0.06). With these outliers removed, there was a positive relationship between $\text{NO}_3\text{-N}$ and PTIA ($p = 0.013$, $r^2 = 0.48$). No streams contained levels of SRP that were above the detectable limit of 0.05 mg L^{-1} . Only Whit, Brew, and AirI contained average TP (0.015, 0.028, and 0.021 mg L^{-1} , respectively) that were above the detectable limit of 0.038 mg L^{-1} .

Macroinvertebrate Community

A total of 99 different macroinvertebrate taxa were identified, with 73 taxa occurring in both seasons. Reference catchments yielded the highest richness (average taxa richness = 34 ± 2 S.E. in the Fall and 32 ± 1 S.E. in the Spring.). The most heavily urbanized catchments yielded the lowest richness (average taxa richness in Barb and AirI <13 taxa). Catchments with urban intensities of 5% PTIA or greater yielded significantly lower annual average total richness than reference catchments ($p = <0.0001$, $t_{18} = 5.44$; Fig. 10a). Additionally, in both Fall and Spring, negative exponential curves best fit the

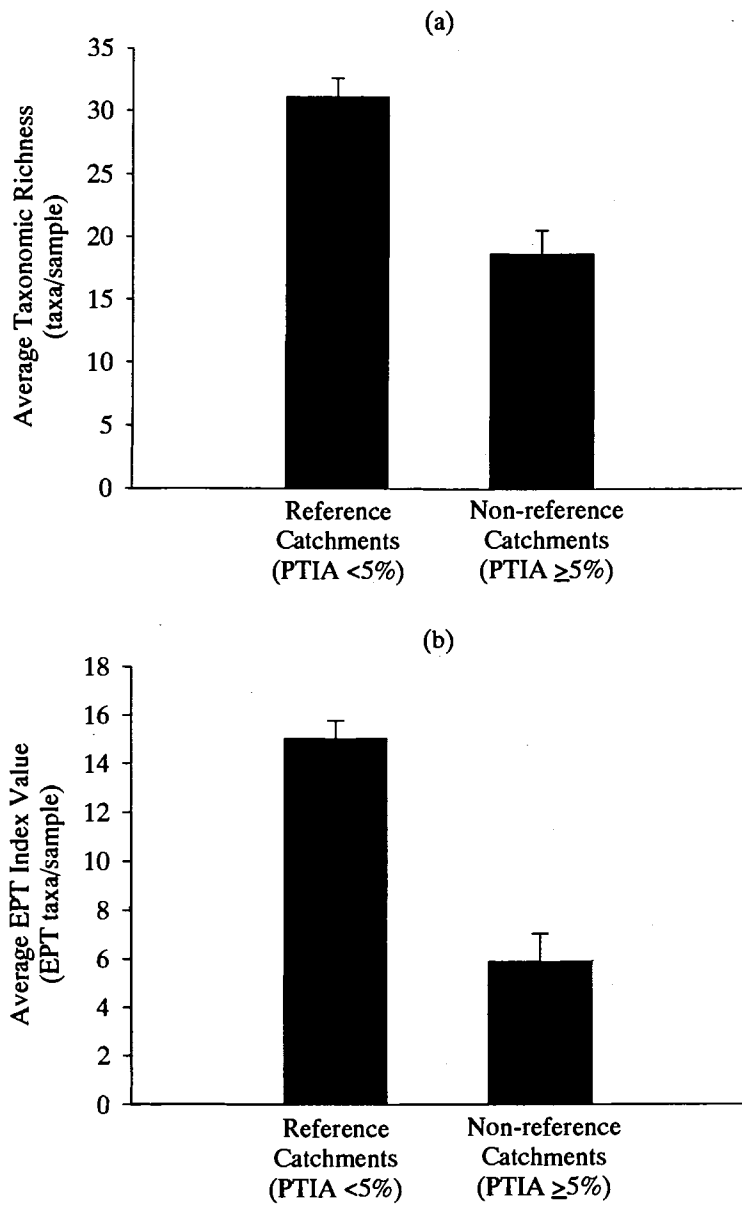


Figure 10. A comparison of (a) annual average taxonomic richness and (b) annual average EPT Index Values of reference and non-reference catchments. N = 8 for reference catchments and n = 12 for non-reference catchments. Error bars represent + 1 S.E.

relationship between average total richness and PTIA across the entire urban gradient ($p = <0.0001$ for both, $r^2 = 0.77$ and 0.81 , respectively). Within the restricted range of 6-27% PTIA (Fig. 11a-b), however, catchments did not show a significant reduction in total richness with increasing PTIA (average total richness within this range = 18 ± 1 S.E. in Fall and 15 ± 1 S.E. in Spring). This indicates the lack of an increasing influence of urbanization on average total richness across this range.

Similar trends were also apparent for the relationship between EPT Index values and PTIA. Reference catchments contained significantly greater numbers of EPT taxa than those catchments with 5% PTIA or greater ($p = <0.0001$, $t_{18} = 5.87$; Fig. 10b). In addition, a reduction in the number of EPT taxa was significantly related to increases in PTIA across the entire urban gradient. Negative exponential curves best fit the trends ($p = <0.0001$ for both, $r^2 = 0.79$ and 0.82 , respectively) with the highest EPT Index values in reference catchments (average EPT richness = 16 ± 1 S.E. in the Fall and 14 ± 1 S.E. in the Spring) and lowest in AirI (average EPT richness = 1 ± 0 in the Fall and 0 ± 0 in the Spring). As with total richness, catchments within the restricted range of 6-27% PTIA (Fig. 11c-d) did not show a significant reduction in the richness of the EPT taxa with increasing levels of PTIA. AirI was unusual with respect to EPT richness, plotting consistently below the range expected for 6-27% PTIA (Fig. 11).

Using criteria based on Plafkin et al. (1989) and average total richness, all catchments with <6% PTIA (including the 8 reference catchments and Jones and Reed both with PTIA = 5%) were rated as “non-impacted”. Jones and Reed were similar to reference catchments with respect to both total and EPT taxa richness (Fig. 11a-d). The majority of catchments with PTIA from 6-27% were rated as “moderately impacted” with

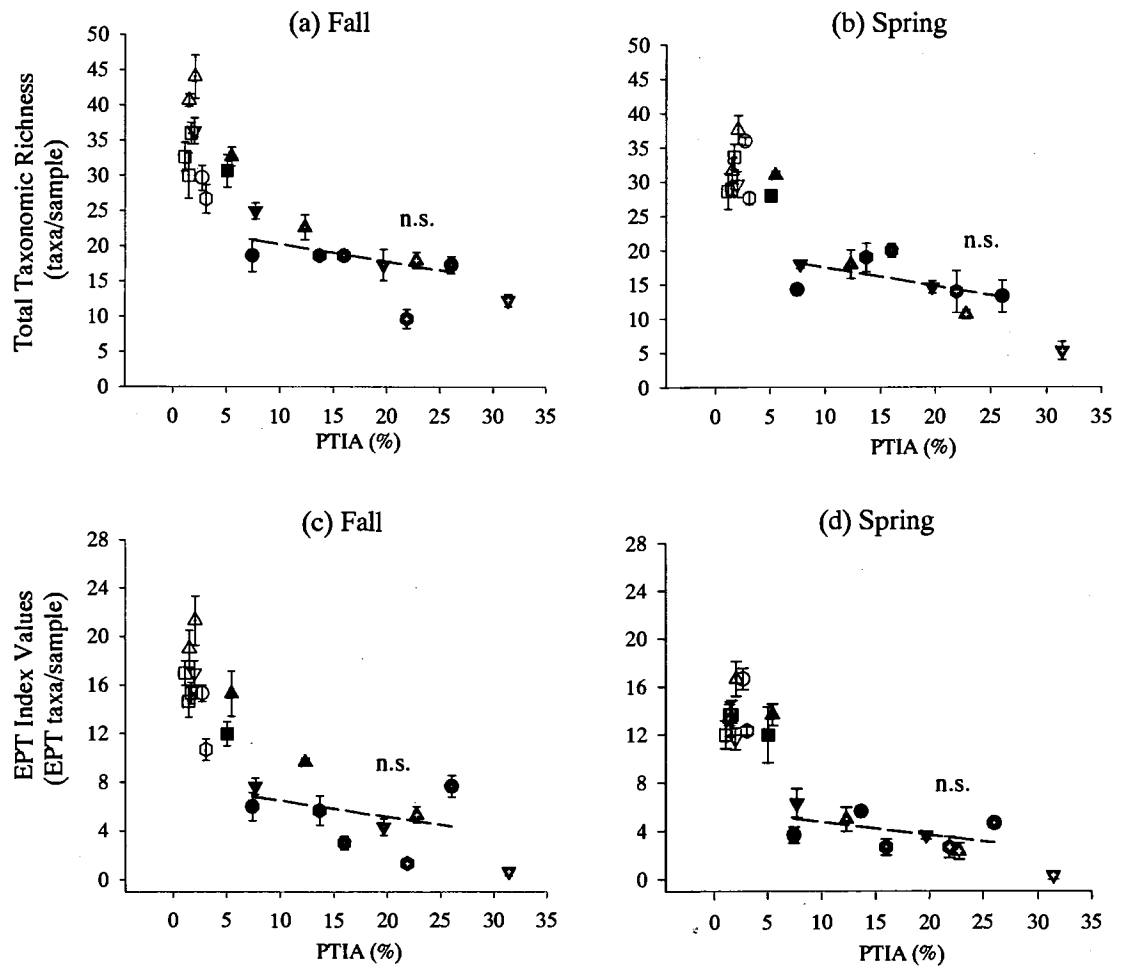


Figure 11. Relationship between PTIA and total richness of the benthic macroinvertebrate community within the (a) Fall and (b) Spring and EPT (Ephemeroptera + Plecoptera + Trichoptera) Index values within the (c) Fall and (d) Spring. Symbols correspond to the legend in Fig. 7. Data points represent 3 samples per reach and error bars indicate ± 1 S.E. Dashed lines indicate non-significant linear regressions (see text).

Long in the Spring and Push in the Fall being “slightly impacted” and Brew in the Spring and Barb in the Fall being “severely impacted”. AirI (the only catchment with >27% PTIA) received a rating of “severely impacted” regardless of season.

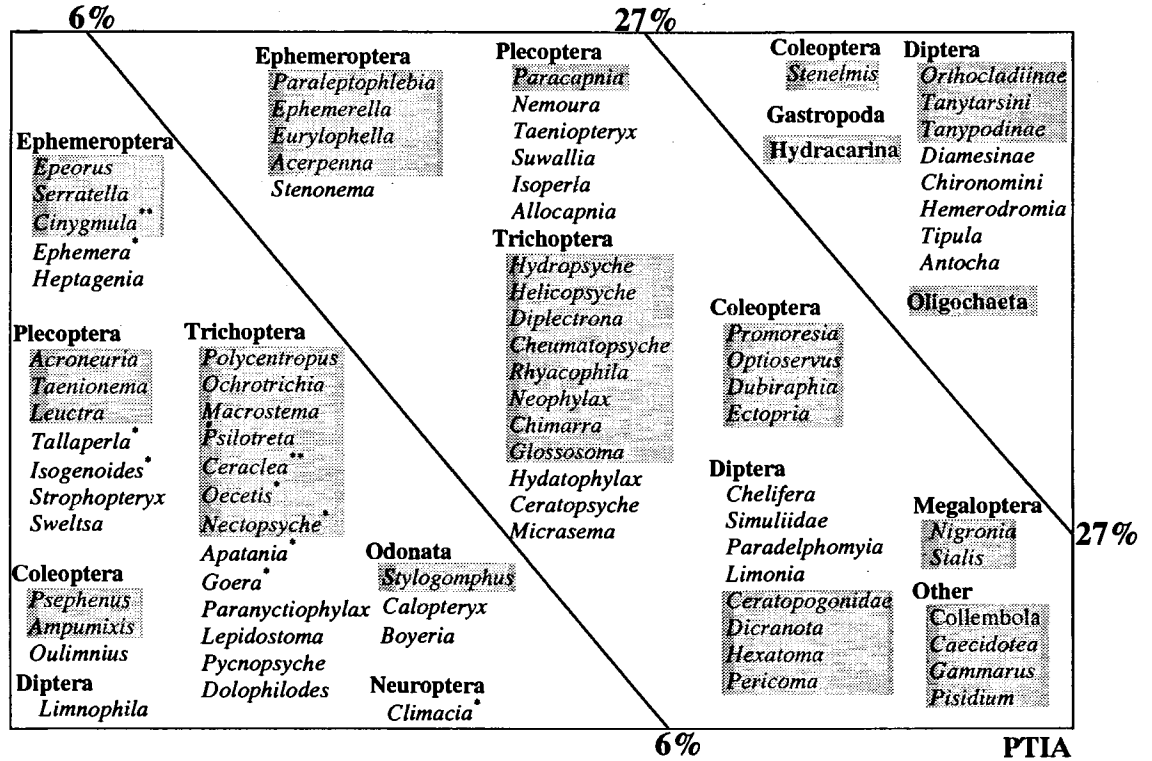
Total richness and EPT Index values were correlated with all physical and water quality variables, with the exception of D_{50} particle sizes. The strongest relationship existed between specific conductance, QHI, and riparian width and EPT Index values in the Fall ($r^2 = 0.72, 0.50, 0.46$, respectively) and Spring ($r^2 = 0.82, 0.46, 0.50$, respectively) and specific conductance and total richness in the Fall ($r^2 = 0.64$) and Spring ($r^2 = 0.74$).

The spatial distribution trends for total and EPT taxa richness are summarized for all sites in Fig. 12(a-b). Within this figure, taxa are present in all sections moving from right to left, (e.g. taxa found in the >27% PTIA range were also found in the 6-27% and <6% ranges of PTIA). Thresholds for the PTIA ranges used within this figure are based on the analyses of the average total richness and average EPT Index values described above. Shaded blocks indicate taxa that were present in the same PTIA range in both seasons. Unique taxa, defined as present in only one sample within a season, are indicated by the “*” notation. Although taxa found to be unique varied seasonally, those catchments in the <6% PTIA range consistently contained 10 unique taxa as opposed to the lower number present in the 6-27% range (two in the Fall and one in the Spring), and the absence of any unique taxa within the >27% range (AirI).

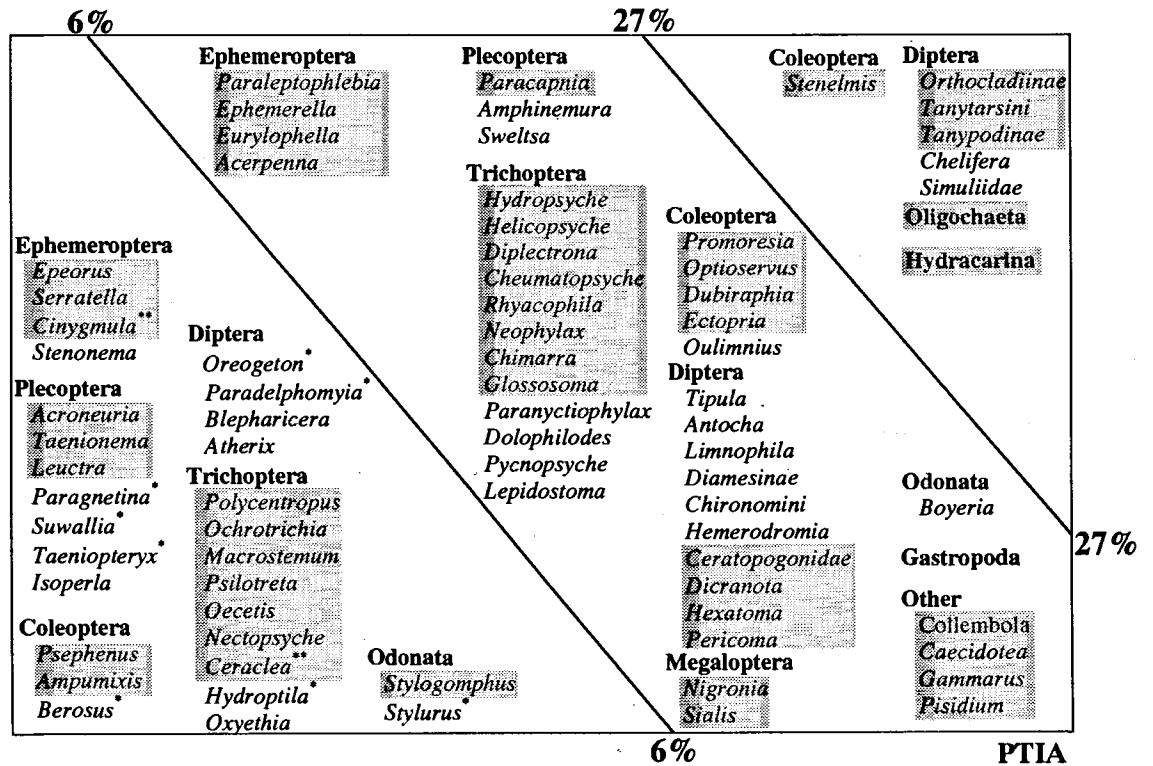
The average total density of macroinvertebrates in the Fall ranged from a maximum of 3534 organisms per sample in Piper to a minimum of 198 in Brew, and densities were negatively related to PTIA ($p = <0.0001, r^2 = 0.36$; Fig. 13a). The average

Figure 12. Presence/absence schemata for all taxa found within the benthic macroinvertebrate communities within the (a) Fall and (b) Spring. Taxa are divided into ranges based on the PTIA values of catchments within which they occurred (PTIA = 0-6%, >6-27% , and >27%). Taxa are present in all preceding ranges moving from left to right. Shading indicates taxa found within the same PTIA ranges within both seasons. The following taxa were not present within continuous ranges of PTIA and were not included in the figure: *Atherix* (Fall) and *Micrasema* (Spring) found only in the <6% and >27% ranges; *Ancronyx*, *Pilaria*, and *Pedicia* (Fall), and *Macronychus* (Spring), found only in the 6%-27% range; *Hyallega* (Fall) found only in the 6%-27% and >27% ranges. Single asterisks indicate taxa found to be unique and double asterisks indicate taxa unique within both seasons. Unique taxa also included *Pedicia* and *Ancronyx* (Fall) and *Macronychus* (Spring) in the 6%-27% PTIA range, not included in the figure as explained above.

(a) Fall Macroinvertebrate Community



(b) Spring Macroinvertebrate Community



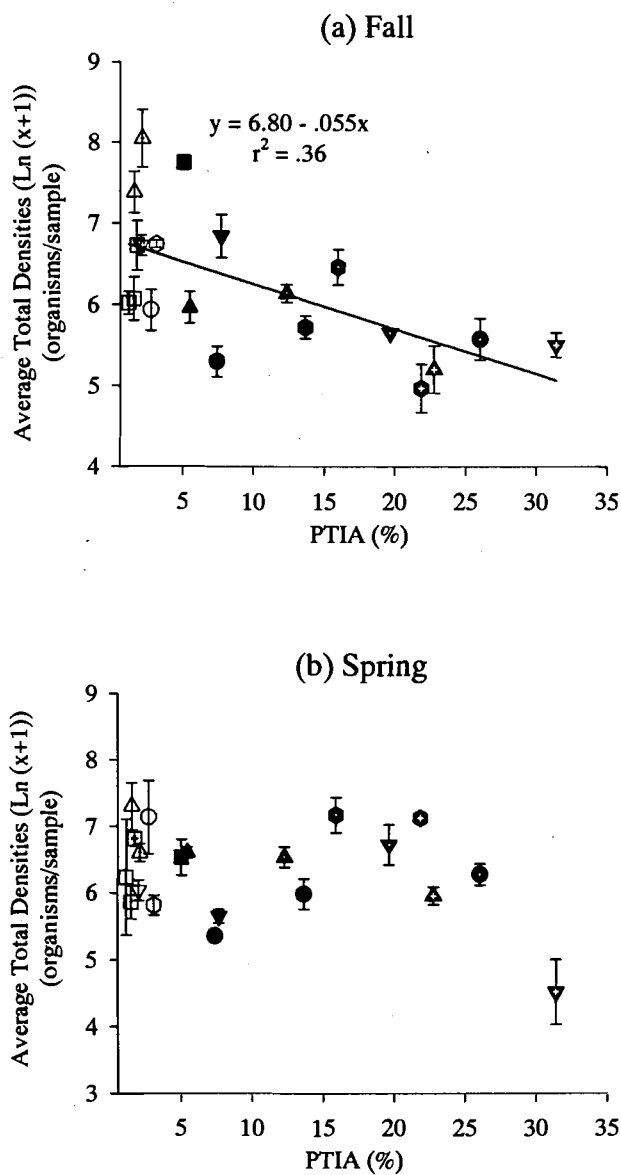


Figure 13. The relationship between PTIA and the average total density of benthic macroinvertebrate samples from study reaches of the (a) Fall and (b) Spring. Symbols correspond to the legend within Fig. 7. Data points represent 3 samples per reach and error bars indicate ± 1 S.E. The solid line within (a) indicates a significant linear regression (see text) with the equation presented within the figure.

total density of macroinvertebrates in the Spring ranged from a maximum of 1253 organisms per sample in Long to a minimum of 52 in AirI, and densities were not significantly related to PTIA in this season (Fig. 13b). The relationship between PTIA and taxon-specific density was examined for common taxa. Common taxa were defined as taxa making up >5% of any sample within either Fall or Spring. A total of 43 taxa within 10 orders were classified as "common". These taxa made up no less than 87% of the total abundance of any sample within either season.

Decreases in the average density of Ephemeroptera, Plecoptera, and Coleoptera were strongly related to increases in PTIA in both seasons (Table 4). Among the common Ephemeroptera taxa, *Stenonema*, *Ephemerella*, and *Paraleptophlebia* (Fall, Spring) and *Serratella* (Fall) showed reductions in average density with increasing PTIA (Table 4). Other common taxa such as *Epeorus* and *Eurylophella* (Fall, Spring) and *Serratella* (Spring) were restricted largely to the reference sites and Jones and Reed (Fig. 14) as were the vast majority of the remaining Ephemeroptera taxa, preventing regression analysis. *Acerpenna* was consistently the only ephemeropteran taxon present in relatively high densities in catchments with >6% PTIA (Fig. 14).

Among the common Plecoptera, the average densities of *Sweltsa* (Fall, Spring), *Leuctra* (Spring), and *Paracapnia* and *Taenionema* (Fall) were negatively related to PTIA (Table 4). The average densities of *Isoperla* and *Allocapnia* were not related to PTIA within either season, nor were densities of *Amphinemura*, a taxon found only in the Spring (Table 5). The Plecoptera exhibited a similar distribution pattern as the Ephemeroptera, with the majority of taxa being found only at the reference sites and Jones and Reed (Fig. 14). *Paracapnia* and *Allocapnia* (Fall) were the only plecopteran

Table 4. Common macroinvertebrate taxa with average densities significantly related to PTIA in the a) Fall and b) Spring. Significance was determined by a $p < 0.05$ from a linear regression of $\ln(x + 1)$ transformed taxa densities and catchment PTIA values. Included within the table are maximum and minimum densities within the study catchments, the r^2 of the linear regression, and the slope of the regression line.

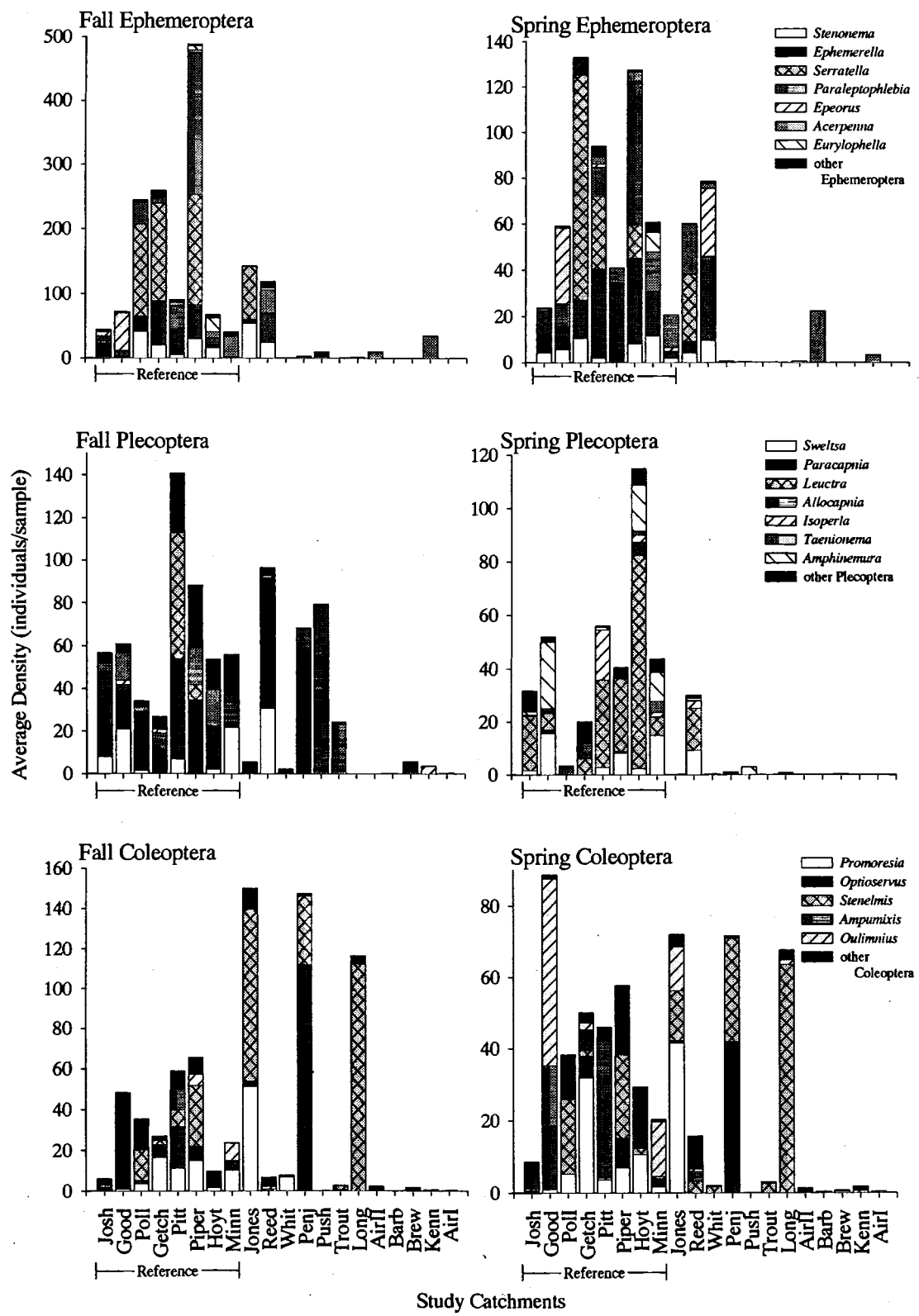
Taxa	Max.		Min.		Density vs. PTIA		
	Density	S.E.	Density	S.E.	r^2	p	slope
Ephemeroptera	488	179	0	0	0.42	*0.002	neg
<i>Stenonema</i>	54	24	0	0	0.29	*0.014	neg
<i>Ephemerella</i>	68	19	0	0	0.47	*0.001	neg
<i>Serratella</i>	170	78	0	0	0.22	*0.038	neg
<i>Paraleptophlebia</i>	222	91	0	0	0.43	*0.002	neg
Plecoptera	141	7	0	0	0.54	*0.000	neg
<i>Sweltsa</i>	31	6	0	0	0.28	*0.016	neg
<i>Paracapnia</i>	59	49	0	0	0.47	*0.001	neg
<i>Taenionema</i>	17	6	0	0	0.21	*0.045	neg
Coleoptera	150	50	0	0	0.45	*0.001	neg
<i>Promoresia</i>	51	14	0	0	0.31	*0.011	neg
Trichoptera							
<i>Rhyacophila</i>	31	9	0	0	0.46	*0.001	neg
Diptera (Chiro)	753	210	56	16	0.24	*0.030	neg
Tanytarsini	157	122	0	0	0.36	*0.006	neg
Chironomini	111	23	0	0	0.68	*0.000	neg
<i>Hexatoma</i>	6	3	0	0	0.25	*0.024	neg
<i>Simuliidae</i>	1501	1069	0	0	0.38	*0.004	neg
Mollusca							
<i>Pisidium</i>	249	220	0	0	0.21	*0.040	neg
Hydracarina	62	5	0	0	0.22	*0.035	neg
Gastropoda	23	11	0	0	0.43	*0.002	pos

Table 4. (continued)

(b) Spring

Taxa	Max.		Min.		Density vs. PTIA		
	Density	S.E.	Density	S.E.	r ²	p	slope
Ephemeroptera	133	41	0	0	0.41	*0.002	neg
<i>Stenonema</i>	12	5	0	0	0.44	*0.002	neg
<i>Ephemerella</i>	39	37	0	0	0.44	*0.001	neg
<i>Paraleptophlebia</i>	63	13	0	0	0.41	*0.002	neg
Plecoptera	115	46	0	0	0.45	*0.001	neg
<i>Sweltsa</i>	15	6	0	0	0.30	*0.012	neg
<i>Leuctra</i>	80	40	0	0	0.41	*0.002	neg
Coleoptera	89	19	0	0	0.41	*0.002	neg
<i>Promoresia</i>	42	12	0	0	0.27	*0.019	neg
<i>Amupumixis</i>	38	14	0	0	0.21	*0.044	neg
Trichoptera	346	215	0	0	0.34	*0.007	neg
<i>Rhyacophila</i>	21	12	0	0	0.49	*0.001	neg
Diptera							
Tanytarsini	156	51	0	0	0.32	*0.009	neg
Chironomini	38	20	0	0	0.28	*0.017	neg
Tanypodinae	89	12	1	0	0.25	*0.025	neg
<i>Hexatoma</i>	11	1	0	0	0.21	*0.040	neg
<i>Simuliidae</i>	763	684	0	0	0.48	*0.001	neg
Mollusca							
<i>Pisidium</i>	92	74	0	0	0.26	*0.021	neg
Hydracarina	24	18	0	0	0.34	*0.007	neg

Figure 14. Comparisons of the densities of selected common taxa within the Ephemeroptera, Plecoptera, Coleoptera, Trichoptera, Diptera and Oligochaeta within the Fall and Spring seasons. Abbreviations found in Table 3. Catchments are in order of increasing PTIA values (see Fig. 4). Stacked bars reflect the density of selected common taxa as found in the legends and maximum bar height reflects the average total density of each order within each catchment.



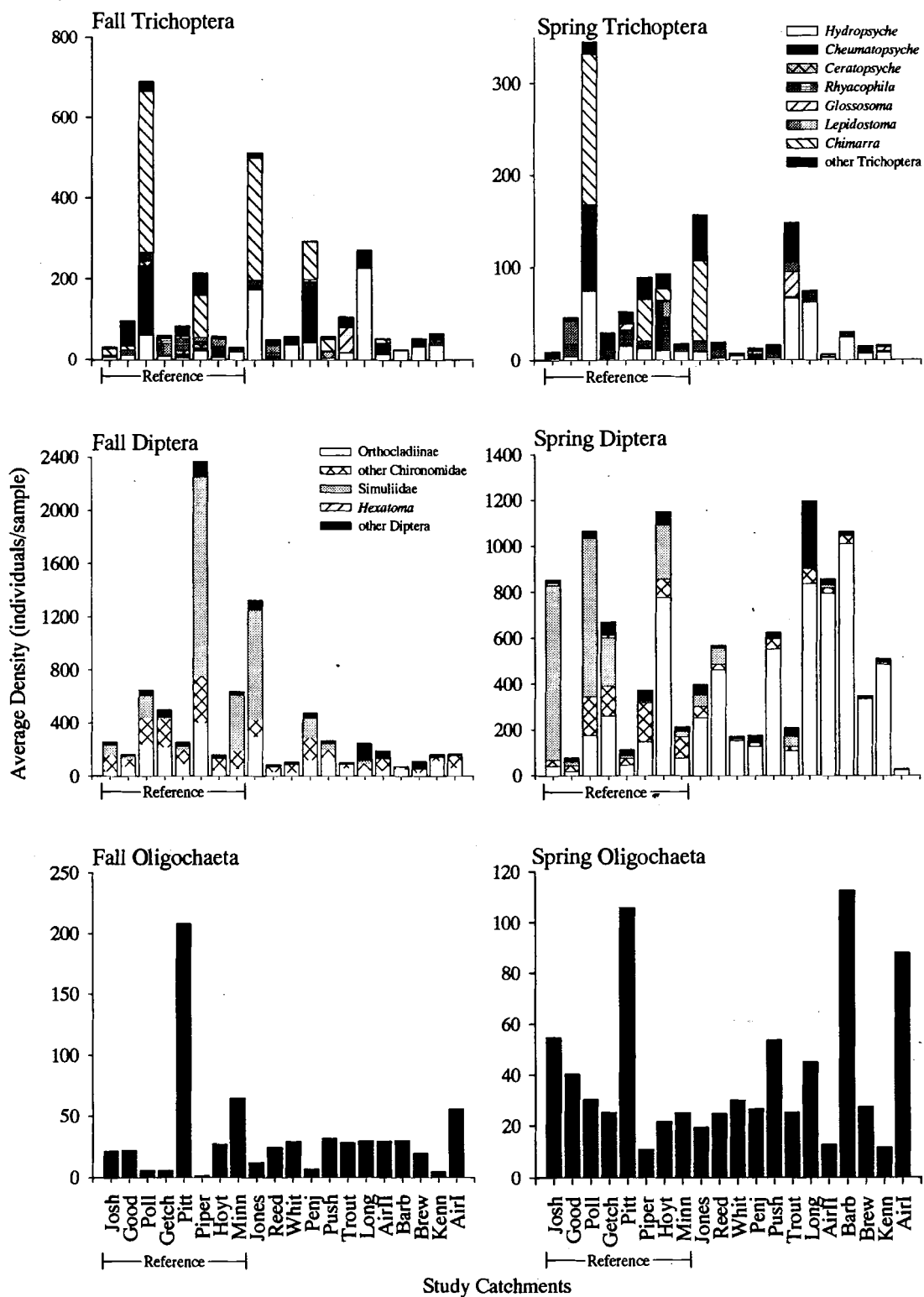


Figure 14. (continued)

Table 5. Common macroinvertebrate taxa with average densities independent of PTIA in the a) Fall and b) Spring. Insignificance was determined by a $p > 0.05$ from a linear regression of $\ln(x + 1)$ transformed taxa densities and catchment PTIA values. Included within the table are maximum and minimum densities within the study catchments, the r^2 of the linear regression, and the slope of the regression line.

(a) Fall							
Taxa	Max.		Min.		Density vs. PTIA		
	Density	S.E.	Density	S.E.	r^2	p	slope
Ephemeroptera							
<i>Epeorus</i>	58	18	0	0	0.11	0.162	pos
<i>Acerpenna</i>	35	9	0	0	0.01	0.699	neg
<i>Eurylophella</i>	21	17	0	0	0.08	0.222	neg
Plecoptera							
<i>Leuctra</i>	59	24	0	0	0.09	0.204	neg
<i>Allocapnia</i>	79	25	0	0	0.05	0.341	neg
<i>Isoperla</i>	4	1	0	0	0.01	0.750	pos
Coleoptera							
<i>Amupumixis</i>	10	10	0	0	0.06	0.294	neg
<i>Optioservus</i>	123	39	0	0	0.18	0.062	neg
<i>Oulimnius</i>	9	4	0	0	0.09	0.200	neg
<i>Stenelmis</i>	123	56	0	0	0.09	0.207	neg
Crustacea							
<i>Caecidotea</i>	27	13	0	0	0.08	0.235	pos
<i>Gammarus</i>	37	9	0	0	0.01	0.678	pos
Trichoptera							
<i>Hydropsyche</i>	227	61	0	0	0.01	0.765	neg
<i>Diplectrona</i>	20	6	0	0	0.03	0.460	pos
<i>Cheumatopsyche</i>	138	69	0	0	0.06	0.360	neg
<i>Lepidostoma</i>	17	11	0	0	0.11	0.157	neg
<i>Dolophilodes</i>	53	51	0	0	0.15	0.092	neg
<i>Glossosoma</i>	60	21	0	0	0.05	0.327	neg
<i>Ceratopsyche</i>	14	9	0	0	0.14	0.106	neg
<i>Chimarra</i>	389	128	0	0	0.14	0.104	neg
Diptera (Total)							
Orthoclaadiinae	398	134	27	15	0.12	0.129	neg
Diamesinae	105	9	0	0	0.10	0.165	pos
Tanypodinae	140	47	2	1	0.02	0.527	neg
<i>Antocha</i>	93	10	0	0	0.08	0.215	pos
<i>Hemerodromia</i>	76	34	0	0	0.01	0.755	neg
<i>Chelifera</i>	17	16	0	0	0.01	0.659	pos
Oligochaeta	28	19	1	1	0.07	0.277	pos

Table 5. (continued)

(b) Spring

Taxa	Max.		Min.		Density vs. PTIA		
	Density	S.E.	Density	S.E.	r ²	p	slope
Ephemeroptera							
<i>Epeorus</i>	33	2	0	0	0.12	0.142	neg
<i>Acerpenna</i>	22	18	0	0	0.02	0.530	neg
<i>Serratella</i>	98	43	0	0	0.18	0.063	neg
<i>Eurylophella</i>	8	6	0	0	0.07	0.254	neg
Plecoptera							
<i>Paracapnia</i>	1	1	0	0	0.07	0.248	neg
<i>Allocapnia</i>	6	3	0	0	0.19	0.056	neg
<i>Taenionema</i>	4	3	0	0	0.07	0.260	neg
<i>Isoperla</i>	19	6	0	0	0.13	0.115	neg
<i>Amphinemura</i>	25	12	0	0	0.18	0.063	neg
Coleoptera							
<i>Optioservus</i>	42	9	0	0	0.15	0.094	neg
<i>Oulimnius</i>	52	27	0	0	0.12	0.130	neg
<i>Stenelmis</i>	64	9	0	0	0.08	0.240	neg
Crustacea							
<i>Caecidotea</i>	37	21	0	0	0.09	0.199	pos
<i>Gammarus</i>	25	9	0	0	0.03	0.460	pos
Tricoptera							
<i>Hydropsyche</i>	75	53	0	0	0.00	0.970	neg
<i>Diplectrona</i>	12	6	0	0	0.01	0.715	pos
<i>Cheumatopsyche</i>	72	30	0	0	0.08	0.213	neg
<i>Lepidostoma</i>	24	21	0	0	0.04	0.401	neg
<i>Dolophilodes</i>	1	1	0	0	0.13	0.126	neg
<i>Glossosoma</i>	27	8	0	0	0.00	0.096	pos
<i>Chimarra</i>	163	122	0	0	0.19	0.056	neg
Diptera (Total)	1196	347	29	5	0.00	0.802	neg
Diptera (Chiro)	1046	103	28	5	0.02	0.603	pos
Orthoclaadiinae	1010	98	20	6	0.00	0.799	pos
Diamesinae	13	8	0	0	0.04	0.396	pos
<i>Hemerodromia</i>	36	10	0	0	0.12	0.139	neg
<i>Chelifera</i>	48	10	0	0	0.00	0.827	neg
<i>Antocha</i>	202	29	0	0	0.01	0.734	pos
Oligochaeta	122	19	11	4	0.19	0.056	pos
Gastropoda	4	1	0	0	0.02	0.531	neg

taxa to persist across the urban gradient in relatively high density (Fig. 14). *Isoperla* and *Amphinemura* occurred sporadically across the entire urban gradient (Fig. 14).

Among the common Coleoptera, *Promoresia* (Fall, Spring) and *Ampumixis* (Spring), a taxon usually associated with streams in the western U.S.A. (Merritt & Cummins 1996), showed reductions in average density that were significantly related to PTIA (Table 4). *Oulimnius* was largely restricted to reference catchments and Jones and Reed (Fig. 14). The densities of both *Optioservus* and *Stenelmis* were not significantly related to PTIA, and made up the majority of Coleoptera within catchments with >6% PTIA range (Fig. 14).

The distribution of Trichoptera with respect to PTIA was variable. Decreases in the average density of Trichoptera were found to be strongly related to PTIA in Fall, while no relationship was apparent during Spring (Table 4, 5). In contrast with the Ephemeroptera and Plecoptera, relatively high densities of Trichoptera persisted across the urban gradient (Fig. 14). At the genus level, only the average density of *Rhyacophila* was significantly and negatively related to PTIA (Table 4). *Chimarra* occurred with high density only in the reference catchments and Jones and Reed (Fig. 14). With the exception of *Hydropsyche* and *Cheumatopsyche*, the remaining common trichopteran taxa were found only sporadically and failed to conform to any pattern related to PTIA (Table 5, Fig. 14). Both *Hydropsyche* and *Cheumatopsyche* occurred in relatively high densities across the urban gradient. These taxa comprised the majority of Trichoptera found within most catchments (8 Fall, 7 Spring) having >6% PTIA (Fig. 14).

The Diptera were found in high densities across the entire urban gradient (Fig. 14) and were the only order that showed a distribution unrelated to PTIA (Table 5). In the

Fall, average density of the Chironomidae was significantly and negatively related with increasing PTIA (Table 4). Within this family, the average density of Tanytarsini and Chironomini (Fall, Spring) and Tanypodinae (Spring) all showed a negative relationship with PTIA (Table 4). However, the average density of the Orthocladiinae was unrelated to PTIA (Table 5). This subfamily was the greatest contributor to both total chironomid and dipteran density (Fig. 14). Exceptions occurred only in catchments with high densities of Simuliidae (Fig. 14). The average densities of the Simuliidae and *Hexatoma* (Tipulidae) had a negative relationship with PTIA (Table 4). The average densities of the remaining common dipteran were apparently unrelated to PTIA (Table 5). Although not common, the order Megaloptera (containing *Nigronia* and *Sialis*) showed reductions in density with increasing PTIA (Fall, $p = 0.012$, $r^2 = 0.30$; Spring, $p = 0.008$, $r^2 = 0.34$).

Oligochaeta were found abundantly at most sites (Fig. 14), however their average densities were unrelated to PTIA (Table 5). The average densities of both *Pisidium* and Hydracarina were negatively related to PTIA (Table 4). Both taxa were almost exclusively found in the reference catchments. *Caecidotea* and *Gammarus*, common crustacean taxa, were found only sporadically and contributed >5% of the total macroinvertebrate density in only one study catchment. The Gastropoda (Fall) was the only common taxa to show an increase in average density with PTIA (Table 4).

Regional Consistency

The relationship of habitat, water quality, and benthic macroinvertebrate community structure to PTIA was consistent among regions. With the exception of the

outliers noted, catchments exhibited similar trends regardless of the regional location of the urban area where they were located (see coded icons in Figs. 8, 9, 11, 13).

Discussion

Macroinvertebrate Community Degradation

Analyses of the benthic macroinvertebrate community structure of the study streams clearly indicated increasing degradation with increasing urban intensity. Reference catchments (<6% PTIA) had richer invertebrate communities with a greater variety of EPT taxa as compared with catchments with >6% PTIA. Reference streams also had higher numbers of unique taxa. EPT taxa, with few important exceptions, were largely absent from catchments with PTIA >6%, where these invertebrate communities were composed largely of pollution-tolerant Trichoptera, Diptera, and Oligochaeta. EPT and Coleoptera taxa that were present in catchments with PTIA >6% (*Acerpenna*, *Paracapnia*, *Allocapnia*, *Cheumatopsyche*, *Hydropsyche*, *Stenelmis*, and *Optioservus*) are documented to be only moderately sensitive to pollution and anthropogenic stresses (Barbour et al. 1999). The macroinvertebrate community of AirI, the highest level of urban intensity in this study and presumably the state of Maine, was composed nearly exclusively of Diptera and Oligochaeta and represented the highest level of degradation as indicated by invertebrate community taxa richness and EPT Index values.

The taxonomic composition of the benthic macroinvertebrate communities was similar between the Fall and Spring with two noteworthy exceptions. First, the Fall samples contained higher densities of *Paracapnia* and *Allocapnia* (Plecoptera). This can be attributed to life history attributes of these taxa. *Paracapnia* and *Allocapnia* are

generally present in highest densities during the Fall coinciding with the input of leaf litter upon which they feed (Merritt & Cummins 1996). Second, the average total density of macroinvertebrates was found to be unrelated to PTIA during Fall. Reductions of average total density in the Spring were, however, significantly related to PTIA.

The trends of decreasing total richness, EPT richness, and density of common taxa with increasing urban intensity, shown in this study, are consistent with results from similar studies (Table 1). The lack of a consistent effect of urban intensity on the total macroinvertebrate density has also been shown in other studies. For example, total density was positively related to urban intensity in Whiting & Goddard (1983) but negatively related in Garie & MacIntosh (1986).

In this correlative study, the mechanism contributing to the degradation of the macroinvertebrate community could not be identified with any certainty. Measurements of physical/morphological condition and water quality did indicate negative influences of increasing PTIA on stream condition. The attributes of the physical/morphological condition and water quality of the study streams, however, offered a single snapshot of within stream conditions at the time of sampling and were not necessarily reflective of long-term conditions present in the streams. Attributes of the invertebrate community, being more integrative measures of stream condition, reflected the influences of stream condition over a longer duration. This variation in the temporal scope of the different stream indicators may account for the lack of consistency between the degradation to the macroinvertebrate community observed and the physical/morphological and water quality attributes measured.

Physical/morphological Influences

Measures of habitat quality and habitat stability did indicate progressive degradation to the physical/morphological component of the study streams across the urban gradient (1% to 31% PTIA). Mean substrate particle size generally decreased with increasing PTIA, likely due to increased erosion and sediment deposition (Richards et al. 1993). Although indicating deteriorating stream condition, habitat quality and stability measures did not provide direct evidence that habitat degradation was the cause of the macroinvertebrate community degradation. Streams with outlying values within the QHI (AirI, Long, AirII, and Brew) or SRISCI (Jones, AirII, and Brew) data did not have noticeably greater or lesser taxa richness or density of common taxa.

The trends of decreasing habitat quality with increasing PTIA observed in this study were, however, consistent with similar studies of the influence of urbanization on in-stream habitat (Booth & Reinelt 1993, Shaver et al. 1994, May et al. 1996, Maxted 1996). Degradation appeared to be not as extensive, however. Booth & Reinelt (1993) evaluated the habitat quality of 140 km of stream channel in Washington state, using a multimetric qualitative index similar to the QHI, and found that nearly all stream reaches influenced by >8% PTIA were “degraded” (e.g. having the lowest habitat quality documented). May et al. (1996) evaluated over 90 stream reaches within a PTIA range of 0-60%, using a modified QHI with categorical ratings of “excellent” (50-60), “good” (40-49), “fair” (25-39), and “poor” (0-24). They found that habitat quality ranged from “excellent” to “poor” within the range of PTIA that corresponded to this study (PTIA = 1-31%). The results of May et al. (1996) and Booth & Reinelt (1993) thus contrast with the results of this study in which all stream reaches had habitat quality ratings exceeding “poor” or “degraded”. A similar contrast occurred for the assessment of channel

stability. Booth & Reinelt (1993) found that all reaches surveyed with >10% PTIA were generally unstable. The majority of reaches surveyed in this study were relatively stable, regardless of PTIA.

This study also failed to detect morphological alterations in stream channel dimensions as a function of PTIA. Changes in channel morphology, such as widening and incision (Leopold 1968, Hammer 1972, Booth 1989, Booth 1990), have been documented to result as channels change to accommodate increased flood flows due to increasing urban development (Hammer 1972, Krug & Goddard 1986, Robinson 1976, Booth 1989). The loss or degradation of the riparian corridor, often associated with urbanization (May 1997; Morse personal observation), can exacerbate this condition. The lack of evidence of changes in channel morphology due to increasing PTIA as might be expected based on observations within other studies (e.g. Leopold 1968, Arnold et al. 1982, Booth 1990) is likely the result of criteria used for catchment selection within this study. As an example, cobble riffles were required to be present within the catchment stream channels to reduce variability within the benthic macroinvertebrate communities that were sampled. Choosing only streams containing cobble riffles could have effectively selected for streams in which habitat degradation was not as great a factor at a given level of PTIA due to the presence of bank material offering increased stability and natural resistance to erosion.

Water Quality Influences

Trends in low-flow water quality, as measured in this study, largely mirrored the trends in the physical/morphological condition of the stream channels. Water quality did show an apparent decline with increasing PTIA, but a catastrophic degradation in water

quality was not indicated; rather, relatively small and incremental changes occurred with increasing urbanization. The worst case scenario for low-flow DO, for example, indicated that pre-dawn DO levels rarely dropped below the 5 mg L^{-1} threshold often used to represent biologically significant DO degradation (Roesner 1982). Although concentrations of $\text{NO}_3\text{-N}$ were positively associated with PTIA, $\text{NO}_3\text{-N}$ levels in this study were generally below concentrations that could be considered significant enrichment. Only Push contained an average $\text{NO}_3\text{-N}$ concentration $>0.6 \text{ mg L}^{-1}$, the concentration identified by the USGS as the national background level in streams (USGS 1999). The results of the water quality sampling conducted within this study are similar to those of studies that concluded that only minimal water quality degradation occurred with increasing urban intensity, as measured during baseflow (Benke 1981, Garie & MacIntosh 1986, Jones et al. 1996). May et al. (1997) reported significant water quality influences only to streams with $>45\%$ PTIA, a level of urban intensity greater than those found within this study.

Even when outlier data were examined, they were ineffective at clearly indicating water quality as a causative factor contributing to the degradation of the invertebrate communities. For example, streams with unusually high specific conductance (Brew), TSS (Long), or $\text{NO}_3\text{-N}$ (Push) did not have unusual invertebrate community attributes. However, lower DO concentrations in Barb could have been responsible for the unusually low level of taxonomic richness measured during the Fall.

Because of the occurrence of high densities of Orthocladiinae and Oligochaeta in the more urbanized catchments, it is possible to hypothesize that increases in TSS led directly to the degradation observed for the invertebrate community structure with increasing PTIA. These taxa are often abundant in depositional habitats (Merritt &

Cummins 1996) and their dominance may indicate a high level of silt accumulation. The coexistence of high densities of the filter-feeding larvae of *Cheumatopsyche* and *Hydropsyche* with these sediment-tolerant taxa, however, indicates that sedimentation may not have been an important factor, although research has shown that *Hydropsyche* can persist under conditions of high siltation (Runde & Hellenthak 2000). The low probability of a siltation effect is supported by the observation that TSS, although increasing across the gradient, were present in concentrations considered to be relatively low (<10 mg L⁻¹; Marsh 1991). Results of the pebble count procedure also indicated that D₁₀ particle size was not significantly related to PTIA.

Duda et al. (1982), Whiting & Clifford (1983), and Pratt et al. (1981) suggested that decreased water quality as a result of inputs of urban runoff containing non-point source (NPS) pollution was the causative agent for the degradation of the macroinvertebrate communities in the urban streams they studied (Table 1). In this study, specific conductance showed the strongest positive relationship of water quality parameters to PTIA ($r^2 = 0.74$). Specific conductance, although an unspecific indicator, can imply NPS pollution by heavy metals, road salt, nutrients, acidity, etc. The persistence of relatively high conductance under low flow conditions could potentially provide evidence that some streams used in this study were chronically influenced by NPS pollution in the absence of storm-flows. Pitt et al. (1995) noted that, in general, the effects of urban runoff on biota are rarely from acute toxicity but rather from long-term chronic exposure. Chronic NPS pollution could contain contaminants responsible for degradation of the invertebrate community. It must be emphasized that significant relationships between total and EPT richness and the majority of both water quality as

well as physical/morphological parameters cannot be considered indicative of causation based on this study. Rather, these stream component attributes were changing in a similar fashion due to the interrelated effects of increasing urban intensity.

PTIA and the Threshold Phenomenon

PTIA is probably best considered a cumulative indicator of the many influences of urbanization on stream ecosystems. This is reflected by the correlation between increasing PTIA and the degradation of the physical/morphological condition, water quality, and invertebrate communities of the study streams.

Previous studies have detected an apparent threshold phenomenon associated with changes to the stream benthic community with increasing PTIA (Klein 1979, Schueler 1994, Arnold & Gibbons 1996), with abrupt changes followed by constant levels of degradation to the community occurring after a given level. The reported threshold level ranges from 5-25% PTIA (Table 1). Results of this study support these previous studies (e.g. Schueler 1994, May 1997) by strongly suggesting a threshold at ~6% PTIA. With few exceptions, total taxa richness and EPT Index values were found to be sharply reduced in catchments with >6% PTIA. Also, no statistical difference was found in the level of total or EPT richness from >6% to <27% PTIA, indicating that there was a constant level of degradation within this range.

This threshold phenomenon is also supported to some degree by patterns of density for selected orders and genera of insects. The density of Ephemeroptera, Plecoptera, and Coleoptera, for example, were considerably higher in the benthic communities of study streams with <6% PTIA as compared with those communities in

streams with >6% PTIA. Additionally, sensitive taxa such as *Seratella* (Ephemeroptera), *Leuctra* (Plecoptera), and *Psilotreta* (Trichoptera) (Barbour et al. 1999) were largely restricted to catchments with <6% PTIA. Only those taxa considered to be moderately sensitive or insensitive to anthropogenic stresses such as *Acerpenna* (Ephemeroptera), *Paracapnia* (Plecoptera), *Hydropsyche* (Trichoptera) and Oligochaeta (Barbour et al. 1999) were found in relatively high densities in catchments with >6% PTIA.

Although not considered reference streams, Jones and Reed yielded invertebrate communities with total and EPT taxa richness and other attributes of community structure that were similar to the reference catchments. For the purposes of this study, reference conditions were defined as catchments with <5% PTIA and with predominately forested cover. Results of the invertebrate community analyses indicate, however, that reference conditions might have been more appropriately defined as forested catchments with <6% PTIA.

Conclusions

In this study of 20 1st and 2nd order streams in Maine, increasing PTIA in the drainage area corresponded with declining habitat quality and decreased diversity and densities of benthic macroinvertebrate taxa. Above a threshold of ~6% PTIA, streams suffered adverse impacts from urbanization, and benthic macroinvertebrate communities exhibited relatively uniform degradation compared with reference streams. As such, PTIA was found to be a very good predictor for distinguishing stream ecosystems under urban stress in central and southern Maine. Based on the results of this study, resource managers within the northeastern U.S.A. may choose to use this indicator to identify streams in catchments threatened by increasing urban intensity. Caution should be used

when estimating PTIA, as PTIA coefficients for commonly found residential intensities reported in the literature were varied considerably from those derived independently in this study (Table 2).

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BIOGRAPHY OF THE AUTHOR

Chandler C. Morse was born in Philadelphia, Pennsylvania and grew up living on the Atlantic coast and in the Midwestern U.S.A. He graduated from Findlay High School, Findlay, Ohio. After attending Temple University and Bowling Green State University, he graduated from The Ohio State University, Columbus, Ohio in 1996 with a Bachelor of Science in Sustainable Resource Management. His focus within this program of study was Natural Resource Policy. After his undergraduate studies, Chandler spent a year on the West Coast, employed in various locations as an environmental educator. In late 1997, Chandler returned east to pursue his graduate studies in the science of environmental issues. Chandler is a candidate for the Master of Science degree in Ecology and Environmental Science from The University of Maine in May, 2001.