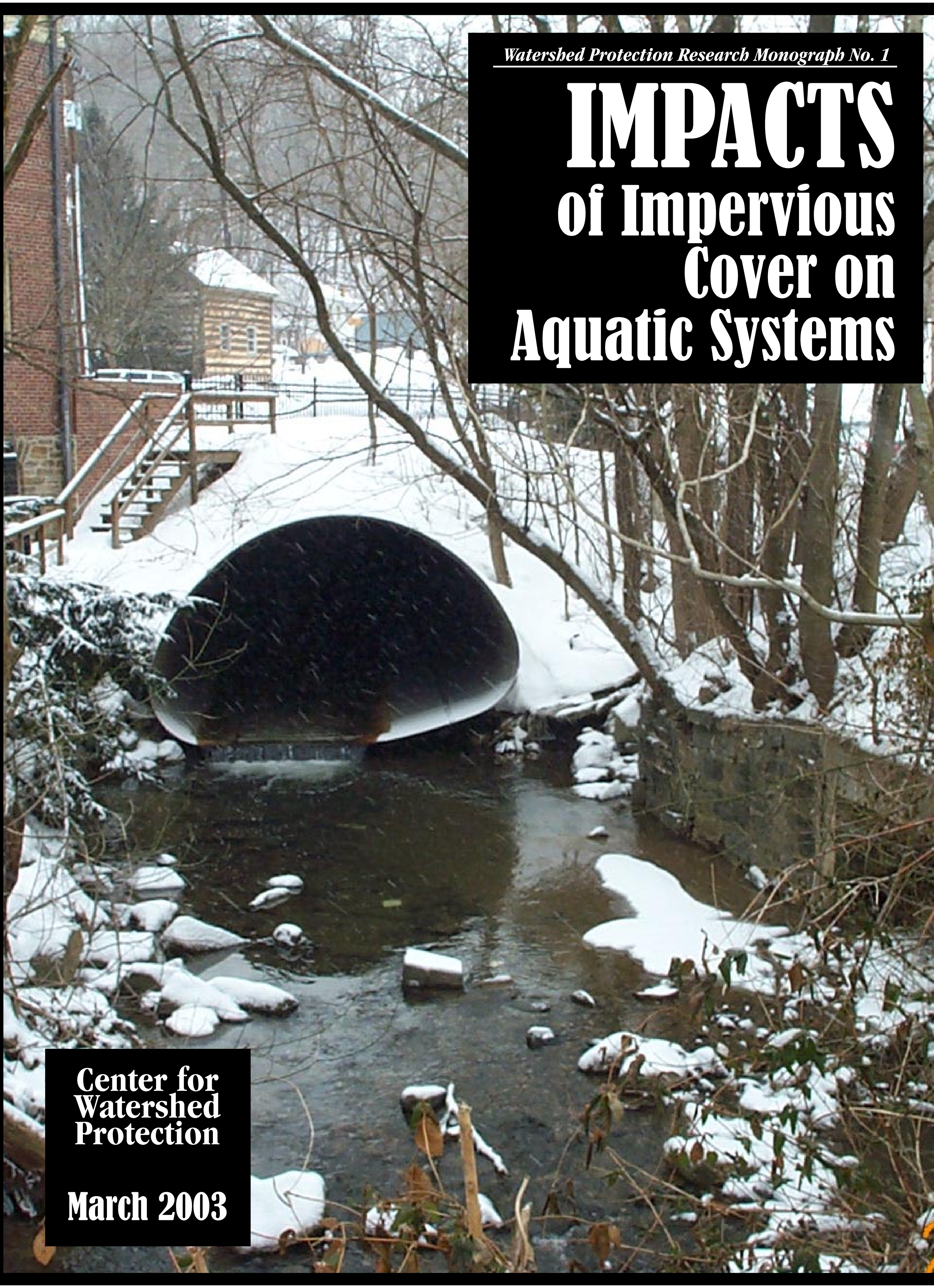
Watershed Protection Research Monograph No. 1

IMPACTS of Impervious Cover on Aquatic Systems



Center for Watershed Protection

March 2003

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March 2003

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Foreword

We are extremely pleased to launch the first edition of a new series called *Watershed Protection Research Monographs*. Each monograph will synthesize emerging research within a major topical area in the practice of watershed protection. The series of periodic monographs will replace our journal *Watershed Protection Techniques,* which lapsed in 2002. We hope this new format will provide watershed managers with the science and perspectives they need to better protect and restore their local watersheds.

This monograph was written to respond to many inquiries from watershed managers and policy makers seeking to understand the scientific basis behind the relationship between impervious cover and the health of aquatic ecosystems. It reviews more than 225 research studies that have explored the impact of impervious cover and other indicators of urbanization on aquatic systems. This report comprehensively reviews the available scientific data on how urbanization influences hydrologic, physical, water quality, and biological indicators of aquatic health, as of late 2002.

Our intention was to organize the available scientific data in a manner that was accessible to watershed leaders, policy-makers and agency staff. In addition, the research itself, which spans dozens of different academic departments and disciplines, was conducted in many different eco-regions, climatic zones, and stream types. In order to communicate across such a wide audience, we have resorted to some simplifications, avoided some important particulars, refrained from some jargon, and tried, wherever possible, to use consistent terminology. Thus, the interpretations and conclusions contained in this document are ours alone, and our readers are encouraged to consult the original sources when in doubt.

We would also like to note that the Center for Watershed Protection and the University of Alabama are currently developing a major national database on stormwater quality. The database will contain nearly 4,000 station-storm events collected by municipalities as part of the U.S. EPA's National Pollutant Discharge Elimination System (NPDES) Phase I Stormwater Permit Program. We anticipate releasing a data report in late 2003 that will provide a much needed update of stormwater event mean concentrations (EMCs).

As of this writing, many research efforts are underway that will further test and refine these relationships (most notably, the U.S. Geological Survey gradients initiative, but also many other local, state and academic efforts). We hope that this report provides a useful summary of the existing science, suggests some directions for new research, and stimulates greater discussion of this important topic in watershed management. We also feel it is time for a major conference or symposium, where this diverse community can join together to discuss methods, findings and the important policy implications of their research.

Acknowledgments

Putting this first research monograph together took a lot of energy, editing and analysis, and many Center staff devoted their time and energy over the last two years to get it done. The project team consisted of Karen Cappiella, Deb Caraco, Samantha Corbin, Heather Holland, Anne Kitchell, Stephanie Linebaugh, Paul Sturm, and Chris Swann. Special thanks are extended to Tiffany Wright, who worked tirelessly to assemble, edit and otherwise polish the final draft.

I am also grateful to Michael Paul of Tetratech, Inc., who graciously provided us with an extensive literature review from his PhD days at the University of Georgia that contained many obscure and hard to find citations. Portions of this monograph were developed as part of a literature review conducted as part of a work assignment for the U.S. EPA Office of Wastewater Management in 2001, which proved indispensable in our efforts. Lastly, I would like to thank the hundreds of scientists who have contributed their time and data to explore and test the relationships between urbanization and aquatic health.

Tom Schueler
Center for Watershed Protection

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Acronyms and Abbreviations

B-IBI	Benthic Index of Biotic Integrity
BOD	Biological Oxygen Demand
BSD	Better Site Design
C-IBI	Combined Index of Biotic Integrity
cfs	cubic feet per second
COD	Chemical Oxygen Demand
CSO	Combined Sewer Overflow
Cu	Copper
DOC	Dissolved Organic Carbon
du/ac	dwelling units per acre
EMC	Event Mean Concentration
EPT	Ephemeroptera, Plecoptera and Trichoptera
FC	Forest Cover
GIS	Geographic Information Systems
IBI	Index of Biotic Integrity
IC	Impervious Cover
ICM	Impervious Cover Model
lbs/ac	pounds per acre
LWD	Large Woody Debris
mg/kg	milligrams per kilogram
mg/l	milligrams per liter (equal to ppm)
MPN	Most Probable Number
MTBE	Methyl Tertiary-Butyl Ether
N	Number of Studies
N/R	data not reported
NO_2	Nitrite
NO_3	Nitrate
NO_x	Nitrogen Oxides
NPDES	National Pollutant Discharge Elimination System
NTU	Nephelometric Turbidity Unit
NURP	National Urban Runoff Program
PAH	Polycylic Aromatic Hydrocarbons
PCB	Polychlorinated Biphenyl
ppb	Parts per billion (equal to ug/l)
ppm	Parts per million (equal to mg/l)
RBP	Rapid Bioassessment Protocol
SLAMM	Source Loading Assessment/ Management Model
SPMD	Semi-Permeable Membrane Device
SSO	Sanitary Sewer Overflow
STP	Stormwater Treatment Practice
TC	Turf Cover
TDS	Total Dissolved Solids
TKN	Total Kjeldhal Nitrogen
TMDL	Total Maximum Daily Load
Total N	Total Nitrogen
Total P	Total Phosphorous
TOC	Total Organic Carbon
TSS	Total Suspended Solids
ug/l	micrograms per liter (equal to ppb)
VMT	Vehicle Miles Traveled
VOC	Volatile Organic Compound
WLF	Water Level Fluctuation
WTP	Wastewater Treatment Plant

Chapter 1: Introduction

This research monograph comprehensively reviews the available scientific data on the impacts of urbanization on small streams and receiving waters. These impacts are generally classified according to one of four broad categories: changes in hydrologic, physical, water quality or biological indicators. More than 225 research studies have documented the adverse impact of urbanization on one or more of these key indicators. In general, most research has focused on smaller watersheds, with drainage areas ranging from a few hundred acres up to ten square miles.

Streams vs. Downstream Receiving Waters

Urban watershed research has traditionally pursued two core themes. One theme has evaluated the direct impact of urbanization on small streams, whereas the second theme has explored the more indirect impact of urbanization on downstream receiving waters, such as rivers, lakes, reservoirs, estuaries and coastal areas. This report is organized to profile recent research progress in both thematic areas and to discuss the implications each poses for urban watershed managers.

When evaluating the direct impact of urbanization on streams, researchers have emphasized hydrologic, physical and biological indicators to define urban stream quality. In recent years, impervious cover (IC) has emerged as a key paradigm to explain and sometimes predict how severely these stream quality indicators change in response to different levels of watershed development. The Center for Watershed Protection has integrated these research findings into a general watershed planning model, known as the impervious cover model (ICM). The ICM predicts that most stream quality indicators decline when watershed IC exceeds 10%, with severe degradation expected beyond 25% IC. In the first part of this review, we critically analyze the scientific basis for the ICM and explore some of its more interesting technical implications.

While many researchers have monitored the quality of stormwater runoff from small watersheds, few have directly linked these pollutants to specific water quality problems within streams (e.g., toxicity, biofouling, eutrophication). Instead, the prevailing view is that stormwater pollutants are a downstream export. That is, they primarily influence downstream receiving water quality. Therefore, researchers have focused on how to estimate stormwater pollutant loads and then determine the water quality response of the rivers, lakes and estuaries that receive them. To be sure, there is an increasing recognition that runoff volume can influence physical and biological indicators within some receiving waters, but only a handful of studies have explored this area. In the second part of this review, we review the impacts of urbanization on downstream receiving waters, primarily from the standpoint of stormwater quality. We also evaluate whether the ICM can be extended to predict water quality in rivers, lakes and estuaries.

This chapter is organized as follows:

1.1 A Review of Recent Urban Stream Research and the ICM
1.2 Impacts of Urbanization on Downstream Receiving Waters
1.3 Implications of the ICM for Watershed Managers

1.1 A Review of Recent Urban Stream Research and the ICM

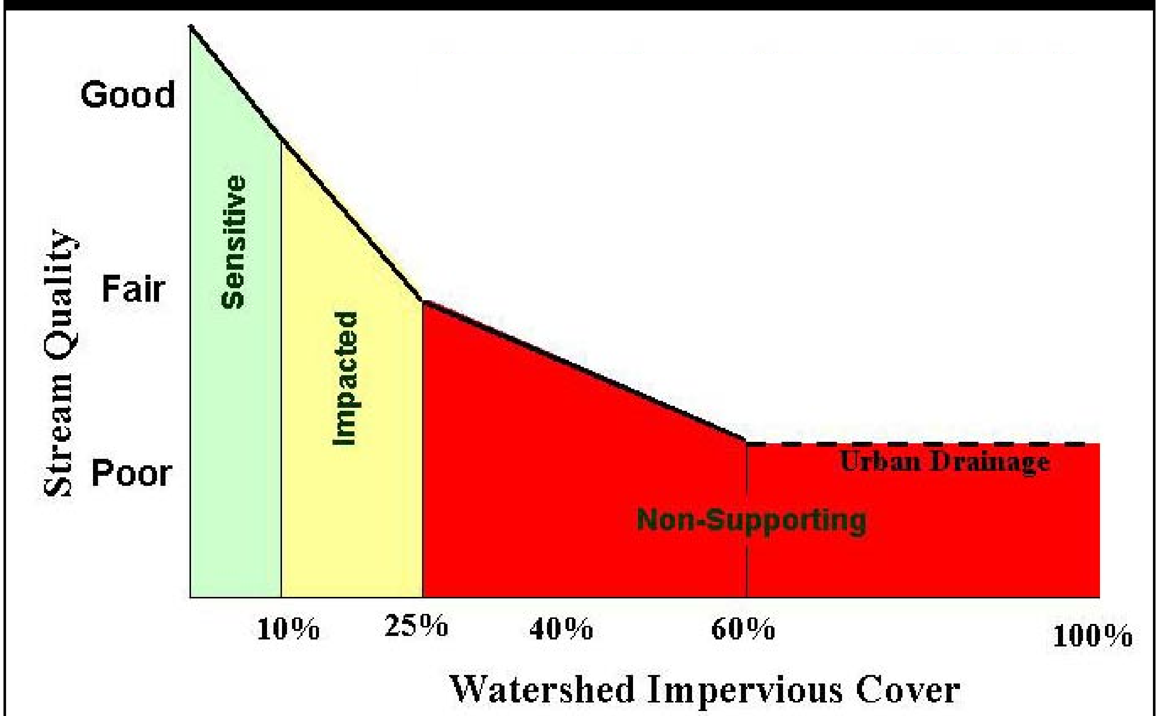
In 1994, the Center published "The Importance of Imperviousness," which outlined the scientific evidence for the relationship between IC and stream quality. At that time, about two dozen research studies documented a reasonably strong relationship between watershed IC and various indicators of stream quality. The research findings were subsequently integrated into the ICM (Schueler, 1994a and CWP, 1998). A brief summary of the basic assumptions of the ICM can be found in Figure 1. The ICM has had a major influence in watershed planning, stream classification and land use regulation in many communities. The ICM is a deceptively simple model that raises extremely complex and profound policy implications for watershed managers.

The ICM has been widely applied in many urban watershed settings for the purposes of small watershed planning, stream classification, and supporting restrictive development regulations and watershed zoning. As such, the ICM has stimulated intense debate among the planning, engineering and scientific communities. This debate is likely to soon spill over into the realm of politics and the courtroom, given its potential implications for local land use and environmental regulation. It is no wonder that the specter of scientific uncertainty is frequently invoked in the ICM debate, given the land use policy issues at stake. In this light, it is helpful to review the current strength of the evidence for and against the ICM.

The ICM is based on the following assumptions and caveats:

- Applies only to 1st, 2nd and 3rd order streams.
- Requires accurate estimates of percent IC, which is defined as the total amount of impervious cover over a subwatershed area.
- Predicts potential rather than actual stream quality. It can and should be expected that some streams will depart from the predictions of the model. For example, monitoring indicators may reveal poor water quality in a stream classified as "sensitive" or a surprisingly high biological diversity

Figure 1: Impervious Cover Model



score in a "non-supporting" one. Consequently, while IC can be used to initially diagnose stream quality, supplemental field monitoring is recommended to actually confirm it.

- Does not predict the precise score of an individual stream quality indicator but rather predicts the average behavior of a group of indicators over a range of IC. Extreme care should be exercised if the ICM is used to predict the fate of individual species (e.g., trout, salmon, mussels).

- "Thresholds" defined as 10 and 25% IC are not sharp "breakpoints," but instead reflect the expected transition of a composite of individual indicators in that range of IC. Thus, it is virtually impossible to distinguish real differences in stream quality indicators within a few percentage points of watershed IC (e.g., 9.9 vs. 10.1%).

- Should only be applied within the ecoregions where it has been tested, including the mid-Atlantic, Northeast, Southeast, Upper Midwest, and Pacific Northwest.

- Has not yet been validated for non-stream conditions (e.g., lakes, reservoirs, aquifers and estuaries).

- Does not currently predict the impact of watershed treatment.

In this section, we review available stream research to answer four questions about the ICM:

1. Does recent stream research still support the basic ICM?
2. What, if any, modifications need to be made to the ICM?
3. To what extent can watershed practices shift the predictions of the ICM?
4. What additional research is needed to test the ICM?

1.1.1 Strength of the Evidence for the ICM

Many researchers have investigated the IC/stream quality relationship in recent years. The Center recently undertook a comprehensive analysis of the literature to assess the scientific basis for the ICM. As of the end of 2002, we discovered more than 225 research studies that measured 26 different urban stream indicators within many regions of North America. We classified the research studies into three basic groups.

The first and most important group consists of studies that directly test the IC/stream quality indicator relationship by monitoring a large population of small watersheds. The second and largest group encompasses secondary studies that indirectly support the ICM by showing significant differences in stream quality indicators between urban and non-urban watersheds. The third and last group of studies includes widely accepted engineering models that explicitly use IC to directly predict stream quality indicators. Examples include engineering models that predict peak discharge or stormwater pollutant loads as a direct function of IC. In most cases, these relationships were derived from prior empirical research.

Table 1 provides a condensed summary of recent urban stream research, which shows the impressive growth in our understanding of urban streams and the watershed factors that influence them. A negative relationship between watershed development and nearly all of the 26 stream quality indicators has been established over many regions and scientific disciplines. About 50 primary studies have tested the IC/stream quality indicator relationship, with the largest number looking at biological indicators of stream health, such as the diversity of aquatic insects or fish. Another 150 or so secondary studies provide evidence that stream quality indicators are significantly different between urban and non-urban watersheds, which lends at least indirect support for the ICM and suggests that additional research to directly test the IC/stream quality indicator

Table 1: The Strength of Evidence: A Review of the Current Research on Urban Stream Indicators						
Stream Quality Indicator	**#**	**IC**	**UN**	**EM**	**RV**	**Notes**
Increased Runoff Volume	2	Y	Y	Y	N	extensive national data
Increased Peak Discharge	7	Y	Y	Y	Y	type of drainage system key
Increased Frequency of Bankfull Flow	2	?	Y	N	N	hard to measure
Diminished Baseflow	8	?	Y	N	Y	inconclusive data
Stream Channel Enlargement	8	Y	Y	N	Y	stream type important
Increased Channel Modification	4	Y	Y	N	?	stream enclosure
Loss of Riparian Continuity	4	Y	Y	N	?	can be affected by buffer
Reduced Large Woody Debris	4	Y	Y	N	?	Pacific NW studies
Decline in Stream Habitat Quality	11	Y	Y	N	?	
Changes in Pool Riffle/Structure	4	Y	Y	N	?	
Reduced Channel Sinuosity	1	?	Y	N	?	straighter channels
Decline in Streambed Quality	2	Y	Y	N	?	embeddedness
Increased Stream Temperature	5	Y	Y	N	?	buffers and ponds also a factor
Increased Road Crossings	3	?	Y	N	?	create fish barriers
Increased Nutrient Load	30+	?	Y	Y	N	higher stormwater EMCs
Increased Sediment Load	30+	?	Y	N	Y	higher EMCs in arid regions
Increased Metals & Hydrocarbons	20+	?	Y	Y	N	related to traffic/VMT
Increased Pesticide Levels	7	?	Y	N	Y	may be related to turf cover
Increased Chloride Levels	5	?	Y	N	Y	related to road density
Violations of Bacteria Standards	9	Y	Y	N	Y	indirect association
Decline in Aquatic Insect Diversity	33	Y	Y	N	N	IBI and EPT
Decline in Fish Diversity	19	Y	Y	N	N	regional IBI differences
Loss of Coldwater Fish Species	6	Y	Y	N	N	trout and salmon
Reduced Fish Spawning	3	Y	Y	N	?	
Decline in Wetland Plant Diversity	2	N	Y	N	?	water level fluctuation
Decline in Amphibian Community	5	Y	Y	N	?	few studies

***#**: total number of all studies that evaluated the indicator for urban watersheds*
***IC**: does balance of studies indicate a progressive change in the indicator as IC increases? Answers: Yes, No or No data (?)*
***UN**: If the answer to IC is no, does the balance of the studies show a change in the indicator from non-urban to urban watersheds? Yes or No*
***EM** Is the IC/stream quality indicator relationship implicitly assumed within the framework of widely accepted engineering models? Yes, No or No models yet exist (?)*
***RV**: If the relationship has been tested in more than one eco-region, does it generally show major differences between ecoregions? Answers: Yes, No, or insufficient data (?)*

relationship is warranted. In some cases, the IC/stream quality indicator relationship is considered so strongly established by historical research that it has been directly incorporated into accepted engineering models. This has been particularly true for hydrological and water quality indicators.

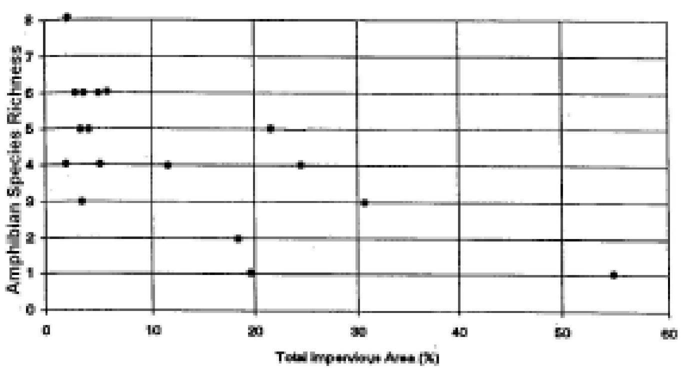
1.1.2 Reinterpretation of the ICM

Although the balance of recent stream research generally supports the ICM, it also offers several important insights for interpreting and applying the ICM, which are discussed next.

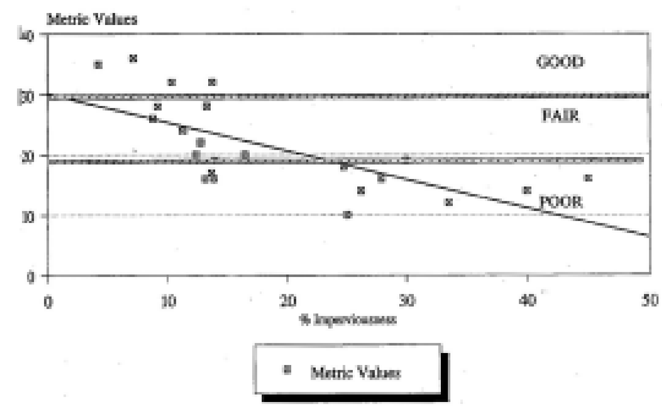
Statistical Variability

Scatter is a common characteristic of most IC/stream quality indicator relationships. In most cases, the overall trend for the indicator is down, but considerable variation exists along the trend line. Often, linear regression equations between IC and individual stream quality indicators produce relatively modest correlation coefficients (reported r^2 of 0.3 to 0.7 are often considered quite strong).

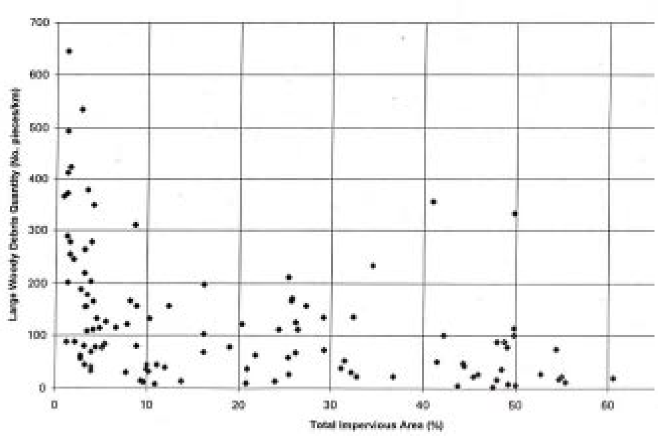
Figure 2 shows typical examples of the IC/stream quality indicator relationship that illustrate the pattern of statistical variability. Variation is always encountered when dealing with urban stream data (particularly so for biological indicators), but several patterns exist that have important implications for watershed managers.



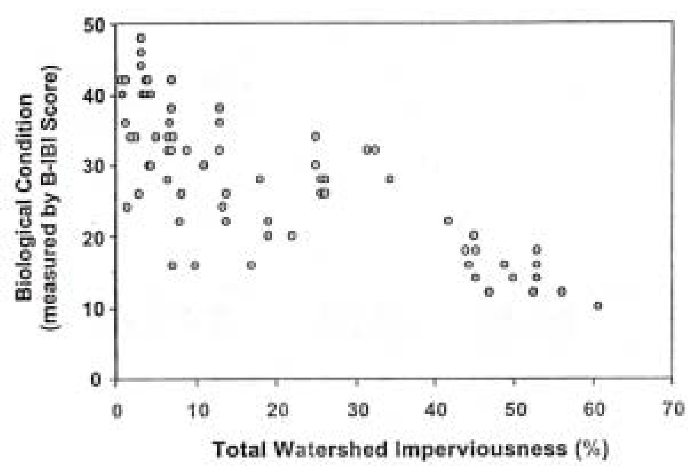
a. Fish IBI vs. IC in Fairfax, VA (Fairfax County, 2001)



b. CPSS vs. IC in Montgomery County, MD (MNCPPC, 2000)



c. Large Woody Debris vs. IC (Booth *et al.*, 1997)



d. Biological Condition vs. Total Watershed IC (Booth, 2000)

Figure 2: Typical Scatter Found in IC/Stream Quality Indicator Research

The first pattern to note is that the greatest scatter in stream quality indicator scores is frequently seen in the range of one to 10% IC. These streams, which are classified as "sensitive" according to the ICM, often exhibit low, moderate or high stream quality indicator scores, as shown in Figure 2. The key interpretation is that sensitive streams have the potential to attain high stream quality indicator scores, but may not always realize this potential.

Quite simply, the influence of IC in the one to 10% range is relatively weak compared to other potential watershed factors, such as percent forest cover, riparian continuity, historical land use, soils, agriculture, acid mine drainage or a host of other stressors. Consequently, watershed managers should never rely on IC alone to classify and manage streams in watersheds with less than 10% IC. Rather, they should evaluate a range of supplemental watershed variables to measure or predict actual stream quality within these lightly developed watersheds.

The second important pattern is that variability in stream quality indicator data is usually dampened when IC exceeds 10%, which presumably reflects the stronger influence of stormwater runoff on stream quality indicators. In particular, the chance that a stream quality indicator will attain a high quality score is sharply diminished at higher IC levels. This trend becomes pronounced within the 10 to 25% IC range and almost inevitable when watershed IC exceeds 25%. Once again, this pattern suggests that IC is a more robust and reliable indicator of overall stream quality beyond the 10% IC threshold.

Other Watershed Variables and the ICM

Several other watershed variables can potentially be included in the ICM. They include forest cover, riparian forest continuity and turf cover.

Forest cover (FC) is clearly the main rival to IC as a useful predictor of stream quality in urban watersheds, at least for humid regions of North America. In some regions, FC is simply the reciprocal of IC. For example, Horner and May (1999) have demonstrated a strong interrelationship between IC and FC for subwatersheds in the Puget Sound region (Figure 3). In other regions, however, "pre-

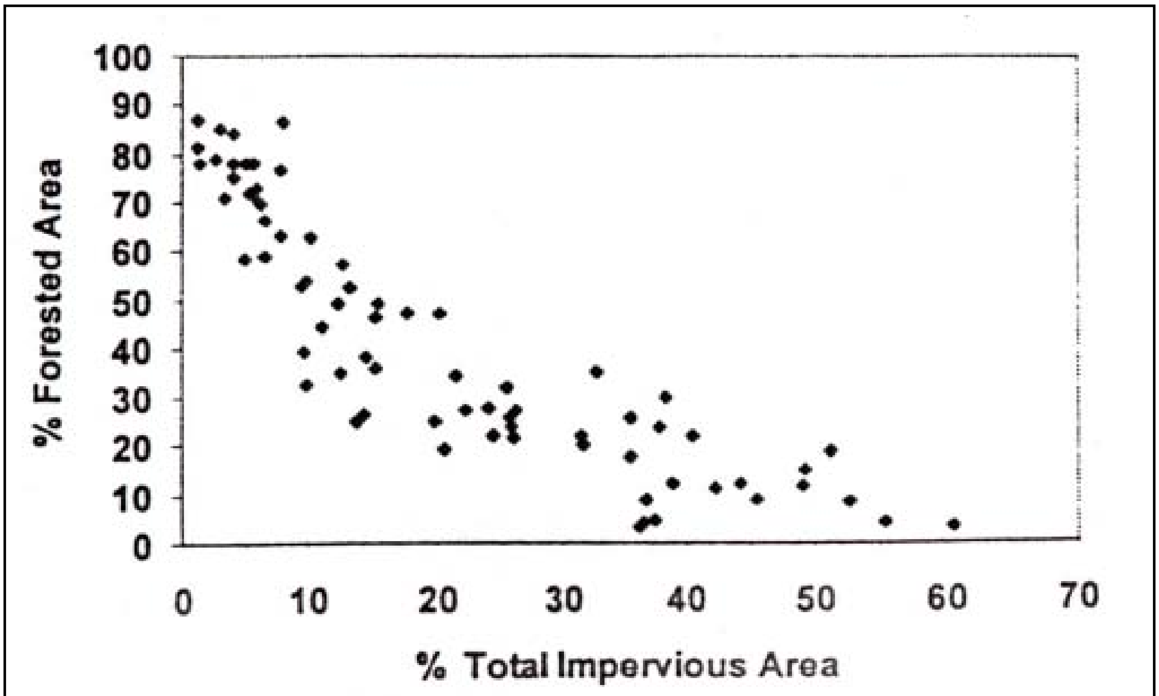


Figure 3: Relationship of IC and FC in Puget Sound Subwatersheds (Horner and May, 1999)

development" land use represents a complex mosaic of crop land, pasture and forest. Therefore, an inverse relationship between FC and IC may not be universal for subwatersheds that have witnessed many cycles of deforestation and cultivation.

It should come as little surprise that the progressive loss of FC has been linked to declining stream quality indicators, given that forested watersheds are often routinely used to define natural reference conditions for streams (Booth, 2000 and Horner *et al.*, 2001). Mature forest is considered to be the main benchmark for defining pre-development hydrology within a subwatershed, as well. Consequently, FC is perhaps the most powerful indicator to predict the quality of streams within the "sensitive" category (zero to 10% IC).

To use an extreme example, one would expect that stream quality indicators would respond quite differently in a subwatershed that had 90% FC compared to one that had 90% crop cover. Indeed, Booth (1991) suggests that stream quality can only be maintained when IC is limited to less than 10% <u>and</u> at least 65% FC is retained within a subwatershed. The key management implication then is that stream health is best managed by simultaneously minimizing the creation of IC and maximizing the preservation of native FC.

FC has also been shown to be useful in predicting the quality of terrestrial variables in a subwatershed. For example, the Mid-Atlantic Integrated Assessment (USEPA, 2000) has documented that watershed FC can reliably predict the diversity of bird, reptile and amphibian communities in the mid-Atlantic region. Moreover, the emerging discipline of landscape ecology provides watershed managers with a strong scientific foundation for deciding where FC should be conserved in a watershed. Conservation plans that protect and connect large forest fragments have been shown to be effective in conserving terrestrial species.

Riparian forest continuity has also shown considerable promise in predicting at least some indicators of stream quality for urban watersheds. Researchers have yet to come up with a standard definition of riparian continuity, but it is usually defined as the proportion of the perennial stream network in a subwatershed that has a fixed width of mature streamside forest. A series of studies indicates that aquatic insect and fish diversity are associated with high levels of riparian continuity (Horner *et al.,* 2001; May *et al.*, 1997; MNCPPC, 2000; Roth *et al.,* 1998). On the other hand, not much evidence has been presented to support the notion that riparian continuity has a strong influence on hydrology or water quality indicators.

One watershed variable that received little attention is the fraction of watershed area maintained in turf cover (TC). Grass often comprises the largest fraction of land area within low-density residential development and could play a significant role in streams that fall within the "impacted" category (10 to 25% IC). Although lawns are pervious, they have sharply different properties than the forests and farmlands they replace (i.e., irrigation, compacted soils, greater runoff, and much higher input of fertilizers and pesticides, etc.). It is interesting to speculate whether the combined area of IC and TC might provide better predictions about stream health than IC area alone, particularly within impacted subwatersheds.

Several other watershed variables might have at least supplemental value in predicting stream quality. They include the presence of extensive wetlands and/or beaverdam complexes in a subwatershed; the dominant form of drainage present in the watershed (tile drains, ditches, swales, curb and gutters, storm drain pipes); the average age of development; and the proximity of sewer lines to the stream. As far as we could discover, none of these variables has been systematically tested in a controlled population of small watersheds. We have observed that these factors could be important in our field investigations and often measure them to provide greater insight into subwatershed behavior.

Lastly, several watershed variables that are closely related to IC have been proposed to predict stream quality. These include population, percent urban land, housing density, road density and other indices of watershed development. As might be expected, they generally track the same trend as IC, but each has some significant technical limitations and/or difficulties in actual planning applications (Brown, 2000).

Individual vs. Multiple Indicators

The ICM does not predict the precise score of individual stream quality indicators, but rather predicts the average behavior of a group of indicators over a range of IC. Extreme care should be exercised if the ICM is used to predict the fate of individual indicators and/or species. This is particularly true for sensitive aquatic species, such as trout, salmon, and freshwater mussels. When researchers have examined the relationship between IC and individual species, they have often discovered lower thresholds for harm. For example, Boward *et al.* (1999) found that brook trout were not found in subwatersheds that had more than 4% IC in Maryland, whereas Horner and May (1999) asserted an 8% threshold for sustaining salmon in Puget Sound streams.

The key point is that if watershed managers want to maintain an individual species, they should be very cautious about adopting the 10% IC threshold. The essential habitat requirements for many sensitive or endangered species are probably determined by the *most sensitive* stream quality indicators, rather than the *average behavior* of all stream quality indicators.

Direct Causality vs. Association

A strong relationship between IC and declining stream quality indicators does not always mean that the IC is directly responsible for the decline. In some cases, however, causality can be demonstrated. For example, increased stormwater runoff volumes are directly caused by the percentage of IC in a subwatershed, although other factors such as conveyance, slope and soils may play a role.

In other cases, the link is much more indirect. For these indicators, IC is merely an index of the cumulative amount of watershed development, and more IC simply means that a greater number of known or unknown pollutant sources or stressors are present. In yet other cases, a causal link appears likely but has not yet been scientifically demonstrated. A good example is the more than 50 studies that have explored how fish or aquatic insect diversity changes in response to IC. While the majority of these studies consistently shows a very strong negative association between IC and biodiversity, they do not really establish which stressor or combination of stressors contributes most to the decline. The widely accepted theory is that IC changes stream hydrology, which degrades stream habitat, and in turn leads to reduced stream biodiversity.

Regional Differences

Currently, the ICM has been largely confirmed within the following regions of North America: the mid-Atlantic, the Northeast, the Southeast, the upper Midwest and the Pacific Northwest. Limited testing in Northern California, the lower Midwest and Central Texas generally agrees with the ICM. The ICM has not been tested in Florida, the Rocky Mountain West, and the Southwest. For a number of reasons, it is not certain if the ICM accurately predicts biological indicators in arid and semiarid climates (Maxted, 1999).

Measuring Impervious Cover

Most researchers have relied on total impervious cover as the basic unit to measure IC at the subwatershed level. The case has repeatedly been made that effective impervious cover is probably a superior metric (e.g., only counting IC that is hydraulically connected to the drainage system). Notwithstanding, most researchers have continued to measure total IC because it is generally quicker and does not require extensive (and often subjective) engineering judgement as to whether it is connected or not. Researchers have used a wide variety of techniques to estimate subwatershed IC, including satellite imagery, analysis of aerial photographs, and derivation from GIS land use layers. Table 2 presents some standard land use/IC relationships that were developed for suburban regions of the Chesapeake Bay.

Table 2: Land Use/IC Relationships for Suburban Areas of the Chesapeake Bay (Cappiella and Brown, 2001)					
Land Use Category	**Sample Number (N)**	**Mean IC (SE)**	**Land Use Category**	**Sample Number (N)**	**Mean IC (SE)**
Agriculture	8	**1.9** – 0.3	Institutional	30	**34.4** – 3.45
Open Urban Land	11	**8.6** – 1.64	Light	20	**53.4** – 2.8
2 Acre Lot Residential	12	**10.6** – 0.65	Commercia	23	**72.2** – 2.0
1 Acre Lot Residential	23	**14.3** – 0.53	Churches	8	**39.9** – 7.8 1
1/2 Acre Lot Residential	20	**21.2** – 0.78	Schools	13	**30.3** – 4.8
1/4 Acre Lot Residential	23	**27.8** – 0.60	Municipals	9	**35.4** – 6.3
1/8 Acre Lot Residential	10	**32.6** – 1.6	Golf	4	**5.0** – 1.7
Townhome Residential	20	**40.9** – 1.39	Cemeteries	3	**8.3** – 3.5
Multifamily Residential	18	**44.4** – 2.0	Parks	4	**12.5** – 0.7

Three points are worth noting. First, it is fair to say that most researchers have spent more quality control effort on their stream quality indicator measurements than on their subwatershed IC estimates. At the current time, no standard protocol exists to estimate subwatershed IC, although Cappiella and Brown (2001) presented a useful method. At best, the different methods used to measure IC make it difficult to compare results from different studies, and at worst, it can introduce an error term of perhaps +/- 10% from the true value within an individual subwatershed. Second, it is important to keep in mind that IC is not constant over time; indeed, major changes in subwatershed IC have been observed within as few as two years. Consequently, it is sound practice to obtain subwatershed IC estimates from the most recent possible mapping data, to ensure that it coincides with stream quality indicator measurements. Lastly, it is important to keep in mind that most suburban and even rural zoning categories exceed 10% IC (see Table 2). Therefore, from a management standpoint, planners should try to project future IC, in order to determine the future stream classification for individual subwatersheds.

1.1.3 Influence of Watershed Treatment Practices on the ICM

The most hotly debated question about the ICM is whether widespread application of watershed practices such as stream buffers or stormwater management can mitigate the impact of IC, thereby allowing greater development density for a given watershed. At this point in time, there are fewer than 10 studies that directly bear on this critical question. Before these are reviewed, it is instructive to look at the difficult technical and scientific issues involved in detecting the effect of watershed treatment, given its enormous implications for land use control and watershed management.

The first tough issue is how to detect the effect of watershed treatment, given the inherent scatter seen in the IC/stream quality indicator relationship. Figure 4 illustrates the “double scatter” problem, based on three different urban stream research studies in Delaware, Maryland and Washington. A quick inspection of the three plots shows how intrinsically hard it is to distinguish the watershed treatment effect. As can be seen, stream quality indicators in subwatersheds with treatment tend to

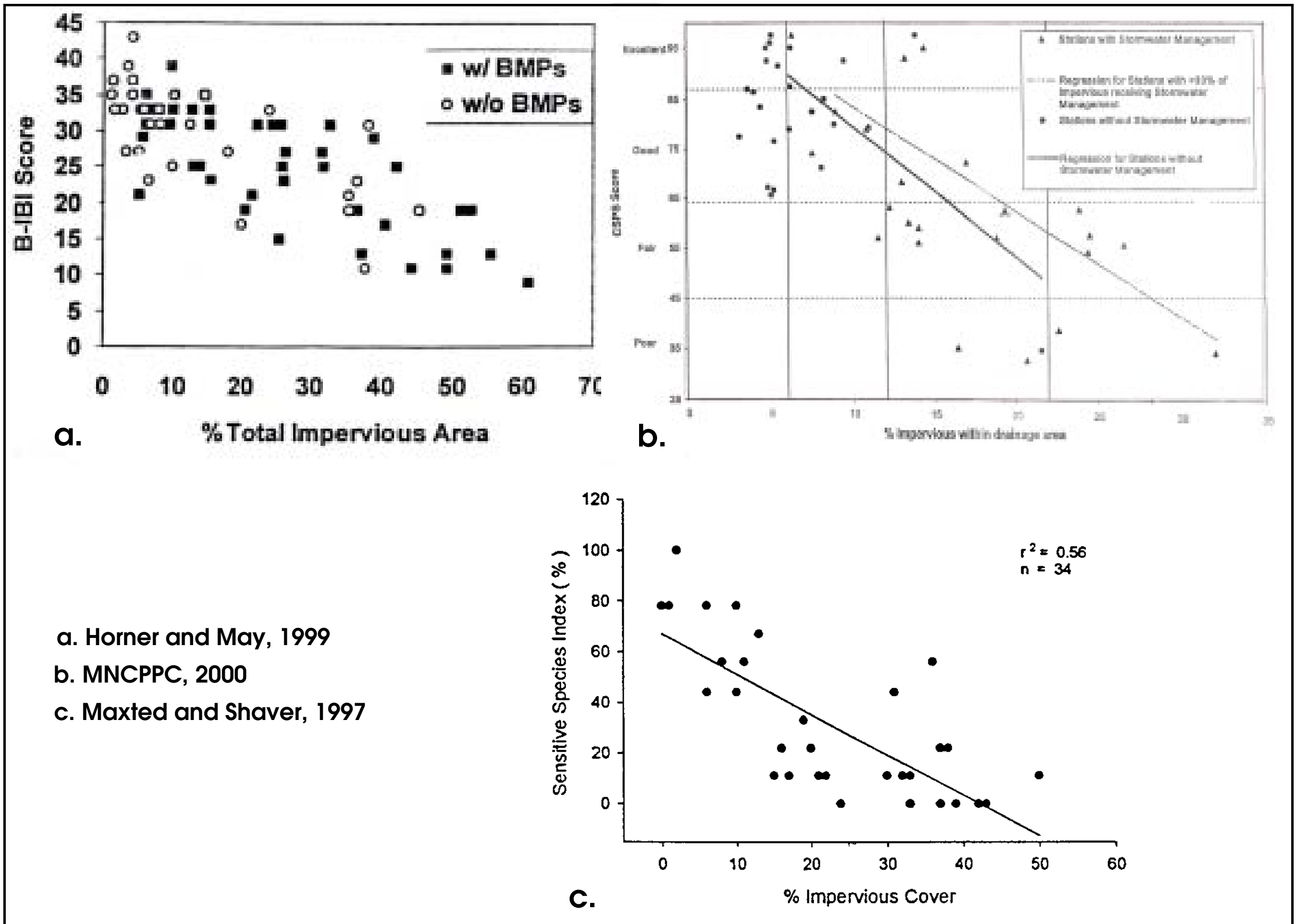


Figure 4: The Double Scatter Problem: Difficulties in Detecting the Effect of Watershed Treatment

overplot those in subwatersheds that lack treatment. While subtle statistical differences may be detected, they are not visibly evident. This suggests that the impact of watershed treatment would need to be extremely dramatic to be detected, given the inherent statistical variability seen in small watersheds (particularly so within the five to 25% IC range where scatter is considerable).

In an ideal world, a watershed study design would look at a controlled population of small urban watersheds that were developed with and without watershed practices to detect the impact of "treatment." In the real world, however, it is impossible to strictly control subwatershed variables. Quite simply, no two subwatersheds are ever alike. Each differs slightly with respect to drainage area, IC, forest cover, riparian continuity, historical land use, and percent watershed treatment. Researchers must also confront other real world issues when designing their watershed treatment experiments.

For example, researchers must carefully choose which indicator or group of indicators will be used to define stream health. IC has a negative influence on 26 stream quality indicators, yet nearly all of the watershed treatment research so far has focused on just a few biological indicators (e.g., aquatic insect or fish diversity) to define stream health. It is conceivable that watershed treatment might have no effect on biological indicators, yet have a positive influence on hydrology, habitat or water quality indicators. At this point, few of these indicators have been systematically tested in the field. It is extremely doubtful that any watershed practice can simultaneously improve or mitigate all 26 stream quality indicators, so researchers must carefully interpret the outcomes of their watershed treatment experiments.

The second issue involves how to quantify watershed treatment. In reality, watershed treatment collectively refers to dozens of practices that are installed at individual development sites in the many years or even decades it takes to fully “build out” a subwatershed. Several researchers have discovered that watershed practices are seldom installed consistently across an entire subwatershed. In some cases, less than a third of the IC in a subwatershed was actually treated by any practice, because development occurred prior to regulations; recent projects were exempted, waived or grandfathered; or practices were inadequately constructed or maintained (Horner and May, 1999 and MNCPPC, 2000).

Even when good coverage is achieved in a watershed, such as the 65 to 90% reported in studies of stormwater ponds (Jones *et al*., 1996; Maxted, 1999; Maxted and Shaver, 1997), it is still quite difficult to quantify the actual quality of treatment. Often, each subwatershed contains its own unique mix of stormwater practices installed over several decades, designed under diverse design criteria, and utilizing widely different stormwater technologies. Given these inconsistencies, researchers will need to develop standard protocols to define the extent and quality of watershed treatment.

Effect of Stormwater Ponds

With this in mind, the effect of stormwater ponds and stream buffers can be discussed. The effect of larger stormwater ponds in mitigating the impacts of IC in small watersheds has received the most scrutiny to date. This is not surprising, since larger ponds often control a large fraction of their contributing subwatershed area (e.g. 100 to 1,000 acres) and are located on the stream itself, therefore lending themselves to easier monitoring. Three studies have evaluated the impact of large stormwater ponds on downstream aquatic insect communities (Jones *et al*., 1996; Maxted and Shaver, 1997; Stribling *et al*., 2001). Each of these studies was conducted in small headwater subwatersheds in the mid-Atlantic Region, and none was able to detect major differences in aquatic insect diversity in streams with or without stormwater ponds.

Four additional studies statistically evaluated the stormwater treatment effect in larger populations of small watersheds with varying degrees of IC (Horner and May, 1999; Horner *et al.,* 2001; Maxted, 1999; MNCPPC, 2000). These studies generally sampled larger watersheds that had many stormwater practices but not necessarily complete watershed coverage. In general, these studies detected a small but positive effect of stormwater treatment relative to aquatic insect diversity. This positive effect was typically seen only in the range of five to 20% IC and was generally undetected beyond about 30% IC. Although each author was hesitant about interpreting his results, all generally agreed that perhaps as much as 5% IC could be added to a subwatershed while maintaining aquatic insect diversity, given effective stormwater treatment. Forest retention and stream buffers were found to be very important, as well. Horner *et al.* (2001) reported a somewhat stronger IC threshold for various species of salmon in Puget Sound streams.

Some might conclude from these initial findings that stormwater ponds have little or no value in maintaining biological diversity in small streams. However, such a conclusion may be premature for several reasons. First, the generation of stormwater ponds that was tested was not explicitly designed to protect stream habitat or to prevent downstream channel erosion, which would presumably promote aquatic diversity. Several states have recently changed their stormwater criteria to require extended detention for the express purpose of preventing downstream channel erosion, and these new criteria may exert a stronger influence on aquatic diversity. Instead, their basic design objective was to maximize pollutant removal, which they did reasonably well.

The second point to stress is that streams with larger stormwater ponds should be considered "regulated streams" (Ward and Stanford, 1979), which have a significantly altered aquatic insect community downstream of the ponds. For example, Galli (1988) has reported that on-stream wet stormwater ponds shift the trophic structure of the aquatic insect community. The insect community above the pond was dominated by shredders, while the insect community below the pond was dominated by scrapers, filterers and collectors. Of particular note, several pollution-sensitive species were eliminated below the pond. Galli reported that changes in stream temperatures, carbon supply and substrate fouling were responsible for the downstream shift in the aquatic insect community. Thus, while it is clear that large stormwater ponds can be expected to have a negative effect on aquatic insect diversity, they could still exert positive influence on other stream quality indicators.

Effect of Stream Buffers

A handful of studies have evaluated biological indicator scores for urban streams that have extensive forest buffers, compared to streams where they were mostly or completely absent (Horner and May, 1999; Horner *et al.,* 2001; May *et al.*, 1997; MNCPPC, 2000; Roth *et al.,* 1998; Steedman, 1988). Biological indicators included various indices of aquatic insect, fish and salmon diversity. Each study sampled a large population of small subwatersheds over a range of IC and derived a quantitative measure to express the continuity, width and forest cover of the riparian buffer network within each subwatershed. Riparian forests were hypothesized to have a positive influence on stream biodiversity, given the direct ways they contribute to stream habitat (e.g., shading, woody debris, leaf litter, bank stability, and organic carbon supply).

All five studies detected a small to moderate positive effect when forested stream buffers were present (frequently defined as at least two-thirds of the stream network with at least 100 feet of stream side forest). The greatest effect was reported by Horner and May (1999) and Horner *et al.* (2001) for salmon streams in the Puget Sound ecoregion. If excellent riparian habitats were preserved, they generally reported that fish diversity could be maintained up to 15% IC, and good aquatic insect diversity could be maintained with as much as 30% IC. Steedman (1988) reported a somewhat smaller effect for Ontario streams. MNCPPC (2000), May *et al.* (1997), and Roth *et al.* (1998) could not find a statistically significant relationship between riparian quality and urban stream quality indicators but did report that most outliers (defined as higher IC subwatersheds with unusually high biological indicator scores) were generally associated with extensive stream side forest.

1.1.4 Recommendations for Further ICM Research

At this point, we recommend three research directions to improve the utility of the ICM for watershed managers. The **first direction** is to expand basic research on the relationship between IC and stream quality indicators that have received little scrutiny. In particular, more work is needed to define the relationship between IC and hydrological and physical indicators such as the following:

- Physical loss or alteration of the stream network
- Stream habitat measures
- Riparian continuity
- Baseflow conditions during dry weather

In addition, more watershed research is needed in ecoregions and physiographic areas where the ICM has not yet been widely tested. Key areas include Florida, arid and semiarid climates, karst areas and mountainous regions. The basic multiple subwatershed monitoring protocol set forth by Schueler (1994a) can be used to investigate IC/stream quality relationships, although it would be wise to measure a wider suite of subwatershed variables beyond IC (e.g., forest cover, turf cover, and riparian continuity).

The **second** research direction is to more clearly define the impact of watershed treatment on stream quality indicators. Based on the insurmountable problems encountered in controlling variation at the subwatershed level, it may be necessary to abandon the multiple watershed or paired watershed sampling approaches that have been used to date. Instead, longitudinal monitoring studies within individual subwatersheds may be a more powerful tool to detect the effect of watershed treatment. These studies could track changes in stream quality indicators in individual subwatersheds over the entire development cycle: pre-development land use, clearing, construction, build out, and post construction. In most cases, longitudinal studies would take five to 10 years to complete, but they would allow watershed managers to measure and control the inherent variability at the subwatershed level and provide a “before and after” test of watershed treatment. Of course, a large population of test subwatersheds would be needed to satisfactorily answer the watershed treatment question.

The **third** research direction is to monitor more non-supporting streams, in order to provide a stronger technical foundation for crafting more realistic urban stream standards and to see how they respond to various watershed restoration treatments. As a general rule, most researchers have been more interested in the behavior of sensitive and impacted streams. The non-supporting stream category spans a wide range of IC, yet we do not really understand how stream quality indicators behave over the entire 25 to 100% IC range.

For example, it would be helpful to establish the IC level at the upper end of the range where streams are essentially transformed into an artificial conveyance system (i.e., become pipes or artificial channels). It would also be interesting to sample more streams near the lower end of the non-supporting category (25 to 35% IC) to detect whether stream quality indicators respond to past watershed treatment or current watershed restoration efforts. For practical reasons, the multiple subwatershed sampling approach is still recommended to characterize indicators in non-supporting streams. However, researchers will need to screen a large number of non-supporting subwatersheds in order to identify a few subwatersheds that are adequate for subsequent sampling (i.e., to control for area, IC, development age, percent watershed treatment, type of conveyance systems, etc.).

1.2 Impacts of Urbanization on Downstream Receiving Waters

In this section, we review the impacts of urbanization on downstream receiving waters, primarily from the standpoint of impacts caused by poor stormwater quality. We begin by looking at the relationship between IC and stormwater pollutant loadings. Next, we discuss the sensitivity of selected downstream receiving waters to stormwater pollutant loads. Lastly, we examine the effect of watershed treatment in reducing stormwater pollutant loads.

1.2.1 Relationship Between Impervious Cover and Stormwater Quality

Urban stormwater runoff contains a wide range of pollutants that can degrade downstream water quality (Table 3). Several generalizations can be supported by the majority of research conducted to date. First, the unit area pollutant load delivered by stormwater runoff to receiving waters increases in direct proportion to watershed IC. This is not altogether surprising, since pollutant load is the product of the average pollutant concentration and stormwater runoff volume. Given that runoff volume increases in direct proportion to IC, pollutant loads must automatically increase when IC increases, as long the average pollutant concentration stays the same (or increases). This relationship is a central assumption in most simple and complex pollutant loading models (Bicknell *et al.*, 1993; Donigian and Huber, 1991; Haith *et al.*, 1992; Novotny and Chester, 1981; NVPDC, 1987; Pitt and Voorhees, 1989).

The second generalization is that stormwater pollutant concentrations are generally similar

Table 3: Summary of Urban Stormwater Pollutant Loads on Quality of Receiving Waters

Pollutants in Urban Stormwater	WQ Impacts To:					Higher Unit Load?	Load a function of IC?	Other Factors Important in Loading
	R	L	E	A	W			
Suspended Sediment	Y	Y	Y	N	Y	Y (ag)	Y	channel erosion
Total Nitrogen	N	N	Y	Y	N	Y (ag)	Y	septic systems
Total Phosphorus	Y	Y	N	N	Y	Y (ag)	Y	tree canopy
Metals	Y	Y	Y	?	N	Y	Y	vehicles
Hydrocarbons	Y	Y	Y	Y	Y	Y	?	related to VMTs and hotspots
Bacteria/Pathogens	Y	Y	Y	N	Y	Y	Y	many sources
Organic Carbon	N	?	?	?	Y	Y	Y	
MTBE	N	N	N	Y	Y	Y	?	roadway, VMTs
Pesticides	?	?	?	?	Y	Y	?	turf/landscaping
Chloride	?	Y	N	Y	Y	Y	?	road density
Trash/Debris	Y	Y	Y	N	?	Y	Y	curb and gutters

Major Water Quality Impacts Reported for:
R = River, L = Lake, E = Estuary, A = Aquifer, W = Surface Water Supply
Higher Unit Area Load? Yes (compared to all land uses) (ag): with exception of cropland
Load a function of IC? Yes, increases proportionally with IC

at the catchment level, regardless of the mix of IC types monitored (e.g., residential, commercial, industrial or highway runoff). Several hundred studies have examined stormwater pollutant concentrations from small urban catchments and have generally found that the variation within a catchment is as great as the variation between catchments. Runoff concentrations tend to be log-normally distributed, and therefore the long term "average" concentration is best expressed by a median value. It should be kept in mind that researchers have discovered sharp differences in pollutant concentrations for smaller, individual components of IC (e.g., rooftops, parking lots, streets, driveways and the like). Since most urban catchments are composed of many kinds of IC, this mosaic quality tempers the variability in long term pollutant concentrations at the catchment or subwatershed scale.

The third generalization is that median concentrations of pollutants in urban runoff are usually higher than in stormwater runoff from most other non-urban land uses. Consequently, the unit area nonpoint pollutant load generated by urban land normally exceeds that of nearly all watershed land uses that it replaces (forest, pasture, cropland, open space — see Table 3). One important exception is cropland, which often produces high unit area sediment and nutrient loads in many regions of the country. In these watersheds, conversion of intensively managed crops to low density residential development may actually result in a slightly decreased sediment or nutrient load. On the other hand, more intensive land development (30% IC or more) will tend to equal or exceed cropland loadings.

The last generalization is that the effect of IC on stormwater pollutant loadings tends to be weakest for subwatersheds in the one to 10% IC range. Numerous studies have suggested that other watershed and regional factors may have a stronger influence, such as the underlying geology, the amount of carbonate rock in the watershed, physiographic region, local soil types, and most important, the relative fraction of forest and crop cover in the subwatershed (Herlihy *et al*., 1998 and Liu *et al.,* 2000). The limited influence of IC on pollutant loads is generally consistent with the finding for hydrologic, habitat and biological indicators over this narrow range of IC. Once again, watershed managers are advised to track other watershed indicators in the sensitive stream category, such as forest or crop cover.

1.2.2 Water Quality Response to Stormwater Pollution

As noted in the previous section, most ICM research has been done on streams, which are directly influenced by increased stormwater. Many managers have wondered whether the ICM also applies to downstream receiving waters, such as lakes, water supply reservoirs and small estuaries. In general, the exact water quality response of downstream receiving waters to increased nonpoint source pollutant loads depends on many factors, including the specific pollutant, the existing loading generated by the converted land use, and the geometry and hydraulics of the receiving water. Table 3 indicates the sensitivity of rivers, lakes, estuaries, aquifers and water supply reservoirs to various stormwater pollutants.

Lakes and the ICM

The water column and sediments of urban lakes are impacted by many stormwater pollutants, including sediment, nutrients, bacteria, metals, hydrocarbons, chlorides, and trash/debris. Of these pollutants, limnologists have always regarded phosphorus as the primary lake management concern, given that more than 80% of urban lakes experience symptoms of eutrophication (CWP, 2001a).

In general, phosphorus export steadily increases as IC is added to a lake watershed, although the precise amount of IC that triggers eutrophication problems is unique to each urban lake. With a little effort, it is possible to calculate the specific IC threshold for an individual lake, given its internal geometry, the size of its contributing watershed, current in-lake phosphorus concentration, degree of watershed treatment, and the desired water quality goals for the lake (CWP, 2001a). As a general rule, most lakes are extremely sensitive to increases in phosphorus loads caused by watershed IC. Exceptions include lakes that are unusually deep and/or have very small drainage area/lake area ratios. In most lakes, however, even a small amount of watershed development will result in an upward shift in trophic status (CWP, 2001a).

Reservoirs and the ICM

While surface water supply reservoirs respond to stormwater pollutant loads in the same general manner as lakes, they are subject to stricter standards because of their uses for drinking water. In particular, water supply reservoirs are particularly sensitive to increased turbidity, pathogens, total organic carbon, chlorides, metals, pesticides and hydrocarbon loads, in addition to phosphorus (Kitchell, 2001). While some pollutants can be removed or reduced through expanded filtering and treatment at drinking water intakes, the most reliable approach is to protect the source waters through watershed protection and treatment.

Consequently, we often recommend that the ICM be used as a "threat index" for most drinking water supplies. Quite simply, if current or future development is expected to exceed 10% IC in the contributing watershed, we recommend that a very aggressive watershed protection strategy be implemented (Kitchell, 2001). In addition, we contend that drinking water quality cannot be sustained once watershed IC exceeds 25% and have yet to find an actual watershed where a drinking water utility has been maintained under these conditions.

Small Tidal Estuaries and Coves and the ICM

The aquatic resources of small tidal estuaries, creeks, and coves are often highly impacted by watershed development and associated activities, such as boating/marinas, wastewater discharge, septic systems, alterations in freshwater flow and wetland degradation and loss. Given the unique impacts of eutrophication on the marine system and stringent water quality standards for shellfish harvesting, the stormwater pollutants of greatest concern in the estuarine water column are nitrogen and fecal coliform bacteria. Metals and hydrocarbons in stormwater runoff can also contaminate bottom sediments, which can prove toxic to local biota (Fortner *et al*., 1996; Fulton *et al*., 1996; Kucklick *et al*., 1997; Lerberg *et al.*, 2000; Sanger *et al.,* 1999; Vernberg *et al.*, 1992).

While numerous studies have demonstrated that physical, hydrologic, water quality and biological indicators differ in urban and non-urban coastal watersheds, only a handful of studies have used watershed IC as an indicator of estuarine health. These studies show significant correlations with IC, although degradation thresholds may not necessarily adhere to the ICM due to tidal dilution and dispersion. Given the limited research, it is not fully clear if the ICM can be applied to coastal systems without modification.

Atmospheric deposition is considered a primary source of nitrogen loading to estuarine watersheds. Consequently, nitrogen loads in urban stormwater are often directly linked to IC. Total nitrogen loads have also been linked to groundwater input, especially from subsurface discharges from septic systems, which are common in low density coastal development (Swann, 2001; Valiela *et al*., 1997; Vernberg *et al*., 1996a). Nitrogen is generally considered to be the limiting nutrient in estuarine systems, and increased loading has been shown to increase algal and phytoplankton biomass and cause shifts in the phytoplankton community and food web structure that may increase the potential for phytoplankton blooms and fish kills (Bowen and Valiela, 2001; Evgenidou *et al*., 1997; Livingston, 1996).

Increased nitrogen loads have been linked to declining seagrass communities, finfish populations, zooplankton reproduction, invertebrate species richness, and shellfish populations (Bowen and Valiela, 2001; Rutkowski *et al*., 1999; Short and Wyllie-Echeverria, 1996; Valiela and Costa, 1988). Multiple studies have shown significant increases in nitrogen loading as watershed land use becomes more urban (Valiela *et al.,* 1997; Vernberg *et al.* 1996a; Wahl *et al.,* 1997). While a few studies link nitrogen loads with building and population density, no study was found that used IC as an indicator of estuarine nitrogen loading.

The second key water quality concern in small estuaries is high fecal coliform levels in stormwater runoff, which can lead to the closure of shellfish beds and swimming beaches. Waterfowl and other wildlife have also been shown to contribute to fecal coliform loading (Wieskel *et al.*, 1996). Recent research has shown that fecal coliform standards are routinely violated during storm events at very low levels of IC in coastal watersheds (Mallin *et al.,* 2001; Vernberg *et al.*, 1996b; Schueler, 1999). Maiolo and Tschetter (1981) found a significant correlation between human population and closed shellfish acreage in North Carolina, and Duda and Cromartie (1982) found greater fecal coliform densities when septic tank density and IC increased, with an approximate threshold at 10% watershed IC.

Recently, Mallin *et al.* (2000) studied five small North Carolina estuaries of different land uses and showed that fecal coliform levels were significantly correlated with watershed population, developed land and IC. Percent IC was the most statistically significant indicator and could explain 95% of the variability in fecal coliform concentrations. They also found that shellfish bed closures were possible in watersheds with less than 10% IC, common in watersheds above 10% IC, and almost certain in watersheds above 20% IC. While higher fecal coliform levels were observed in developed watersheds, salinity, flushing and proximity to pollution sources often resulted in higher concentrations at upstream locations and at high tides (Mallin *et al.*, 1999). While these studies support the ICM, more research is needed to prove the reliability of the ICM in predicting shellfish bed closures based on IC.

Several studies have also investigated the impacts of urbanization on estuarine fish, macrobenthos and shellfish communities. Increased PAH accumulation in oysters, negative effects of growth in juvenile sheepshead minnows, reduced molting efficiency in copepods, and reduced numbers of grass shrimp have all been reported for urban estuaries as compared to forested estuaries (Fulton *et al.,* 1996). Holland *et al. (*1997) reported that the greatest abundance of penaid shrimp and mummichogs was observed in tidal creeks with forested watersheds compared to those with urban cover. Porter *et al.* (1997) found lower grass shrimp abundance in small tidal creeks adjacent to commercial and urban development, as compared to non-urban watersheds.

Lerberg *et al.* (2000) studied small tidal creeks and found that highly urban watersheds (50% IC) had the lowest benthic diversity and abundance as compared to suburban and forested creeks, and benthic communities were numerically dominated by tolerant oligochaetes and polychaetes. Suburban watersheds (15 to 35% IC) also showed signs of degradation and had some pollution tolerant macrobenthos, though not as markedly as urban creeks. Percent abundance of pollution-indicative species showed a marked decline at 30% IC, and the abundance of pollution-sensitive species also significantly correlated with IC (Lerberg *et al*., 2000). Holland *et al*. (1997) reported that the variety and food availability for juvenile fish species was impacted at 15 to 20% IC.

Lastly, a limited amount of research has focused on the direct impact of stormwater runoff on salinity and hypoxia in small tidal creeks. Blood and Smith (1996) compared urban and forested watersheds and found higher salinities in urban watersheds due to the increased number of impoundments. Fluctuations in salinity have been shown to affect shellfish and other aquatic populations (see Vernberg, 1996b). When urban and forested watersheds were compared, Lerberg *et al.* (2000) reported that higher salinity fluctuations occurred most often in developed watersheds; significant correlations with salinity range and IC were also determined. Lerberg *et al.* (2000) also found that the most severe and frequent hypoxia occurred in impacted salt marsh creeks and that dissolved oxygen dynamics in tidal creeks were comparable to dead-end canals common in residential marina-style coastal developments. Suburban watersheds (15 to 35% IC) exhibited signs of degradation and had some pollution-tolerant macrobenthic species, though not to the extent of urban watersheds (50% IC).

In summary, recent research suggests that indicators of coastal watershed health are linked to IC. However, more research is needed to clarify the relationship between IC and estuarine indicators in small tidal estuaries and high salinity creeks.

1.2.3 Effect of Watershed Treatment on Stormwater Quality

Over the past two decades, many communities have invested in watershed protection practices, such as stormwater treatment practices (STPs), stream buffers, and better site design, in order to reduce pollutant loads to receiving waters. In this section, we review the effect of watershed treatment on the quality of stormwater runoff.

Effect of Stormwater Treatment Practices

We cannot directly answer the question as to whether or not stormwater treatment practices can significantly reduce water quality impacts at the watershed level, simply because no controlled monitoring studies have yet been conducted at this scale. Instead, we must rely on more indirect research that has tracked the change in mass or concentration of pollutants as they travel through individual stormwater treatment practices. Thankfully, we have an abundance of these performance studies, with nearly 140 monitoring studies evaluating a diverse range of STPs, including ponds, wetlands, filters, and swales (Winer, 2000).

These studies have generally shown that stormwater practices have at least a moderate ability to remove many pollutants in urban stormwater. Table 4 provides average removal efficiency rates for a range of practices and stormwater pollutants, and Table 5 profiles the mean storm outflow concentrations for various practices. As can be seen, some groups of practices perform better than others in removing certain stormwater pollutants. Consequently, managers need to carefully choose which practices to apply to solve the primary water quality problems within their watersheds.

It is also important to keep in mind that site-based removal rates cannot be extrapolated to the watershed level without significant adjustment. Individual site practices are never implemented perfectly or consistently across a watershed. At least three discount factors need to be considered: bypassed load, treatability and loss of performance over time. For a review on how these discounts are derived, consult Schueler and Caraco (2001). Even under the most optimistic watershed implementation scenarios, overall pollutant reduc-

Table 4: The Effectiveness of Stormwater Treatment Practices in Removing Pollutants - Percent Removal Rate (Winer, 2000)

Practice	N	TSS	TP	OP	TN	NOx	Cu	Zn	Oil/ Grease[1]	Bacteria
Dry Ponds	9	47	19	N/R	25	3.5	26	26	3	44
Wet Ponds	43	80	51	65	33	43	57	66	78	70
Wetlands	36	76	49	48	30	67	40	44	85	78
Filtering Practices[2]	18	86	59	57	38	-14	49	88	84	37
Water Quality Swales	9	81	34	1.0	84	31	51	71	62	-25
Ditches[3]	9	31	-16	N/R	-9.0	24	14	0	N/R	0
Infiltration	6	95	80	85	51	82	N/R	N/R	N/R	N/R

1: Represents data for Oil and Grease and PAH
2: Excludes vertical sand filters
3: Refers to open channel practices not designed for water quality
N/R = Not Reported

Table 5: Median Effluent Concentrations from Stormwater Treatment Practices (mg/l) (Winer, 2000)

Practice	N	TSS	TP	OP	TN	NOx	Cu[1]	Zn[1]
Dry Ponds[2]	3	28	0.18	N/R	0.86	N/R	9.0	98
Wet Ponds	25	17	0.11	0.03	1.3	0.26	5.0	30
Wetlands	19	22	0.20	0.07	1.7	0.36	7.0	31
Filtering Practices[3]	8	11	0.10	0.07	1.1	0.55	9.7	21
Water Quality Swales	7	14	0.19	0.09	1.1	0.35	10	53
Ditches[4]	3	29	0.31	N/R	2.4	0.72	18	32

1. Units for Zn and Cu are micrograms per liter (Fg/l)
2. Data available for Dry Extended Detention Ponds only
3. Excludes vertical sand filters
4. Refers to open channel practices not designed for water quality
N/R = Not Reported

tions by STPs may need to be discounted by at least 30% to account for partial watershed treatment.

Even with discounting, however, it is evident that STPs can achieve enough pollutant reduction to mimic rural background loads for many pollutants, as long as the watershed IC does not exceed 30 to 35%. This capability is illustrated in Figure 5, which shows phosphorus load as a function of IC, with and without stormwater treatment.

Effect of Stream Buffers/Riparian Areas

Forested stream buffers are thought to have very limited capability to remove stormwater pollutants, although virtually no systematic monitoring data exists to test this hypothesis. The major reason cited for their limited removal capacity is that stormwater generated from upland IC has usually concentrated before it reaches the forest buffer and therefore crosses the buffer in a channel, ditch or storm drain pipe. Consequently, the opportunity to filter runoff is lost in many forest buffers in urban watersheds.

Effect of Better Site Design

Better site design (BSD) is a term for nonstructural practices that minimize IC, conserve natural areas and distribute stormwater treatment across individual development sites. BSD is also known by many other names, including conservation development, low-impact development, green infrastructure, and sustainable urban drainage systems. While

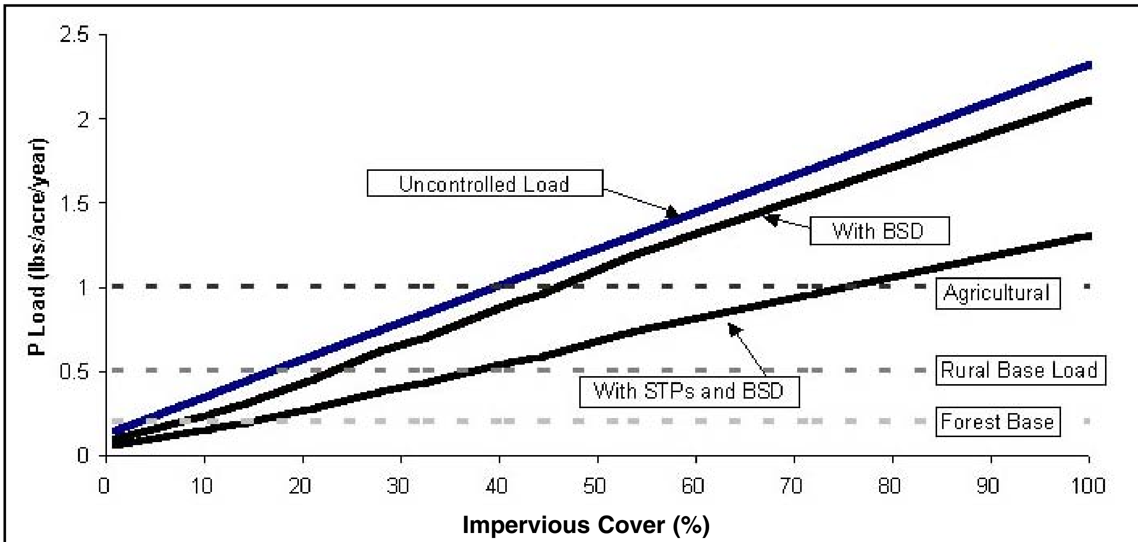


Figure 5: Estimated Phosphorus Load as a Function of Impervious Cover, Discounted Stormwater Treatment and Better Site Design (Schueler and Caraco, 2001)

some maintain that BSD is an alternative to traditional STPs, most consider it to be an important complement to reduce pollutant loads.

While BSD has become popular in recent years, only one controlled research study has evaluated its potential performance, and this is not yet complete (i.e. Jordan Cove, CT).

Indirect estimates of the potential value of BSD to reduce pollutant discharges have been inferred from modeling and redesign analyses (Zielinski, 2000). A typical example is provided in Figure 5, which shows the presumed impact of BSD in reducing phosphorus loadings. As is apparent, BSD appears to be a very effective strategy in the one to 25% IC range, but its benefits diminish beyond that point.

1.3 Implications of the ICM for Watershed Managers

One of the major policy implications of the ICM is that in the absence of watershed treatment, it predicts negative stream impacts at an extremely low intensity of watershed development. To put this in perspective, consider that a watershed zoned for two-acre lot residential development will generally exceed 10% IC, and therefore shift from a sensitive to an impacted stream classification (Cappiella and Brown, 2001). Thus, if a community wants to protect an important water resource or a highly regarded species (such as trout, salmon or an endangered freshwater mussel), the ICM suggests that there is a maximum limit to growth that is not only quite low, but is usually well below the current zoning for many suburban or even rural watersheds. Consequently, the ICM suggests the unpleasant prospect that massive down-zoning, with all of the associated political and legal carnage involving property rights and economic development, may be required to maintain stream quality.

It is not surprising, then, that the ICM debate has quickly shifted to the issue of whether or not watershed treatment practices can provide adequate mitigation for IC. How much relief can be expected from stream buffers, stormwater ponds, and other watershed practices, which might allow greater development density within a given watershed? Only a limited amount of research has addressed this question, and the early results are not reassuring (reviewed in section 1.1.3). At this early stage, researchers are still having trouble detecting the impact of watershed treatment, much less defining it. As noted earlier, both watershed research techniques and practice implementation need to be greatly improved if we ever expect to get a scientifically defensible answer to this crucial question. Until then, managers should be extremely cautious in setting high expectations for how much watershed treatment can mitigate IC.

1.3.1 Management of Non-Supporting Streams

Most researchers acknowledge that streams with more than 25% IC in their watersheds cannot support their designated uses or attain water quality standards and are severely degraded from a physical and biological standpoint. As a consequence, many of these streams are listed for non-attainment under the Clean Water Act and are subject to Total Maximum Daily Load (TMDL) regulations. Communities that have streams within this regulatory class must prepare implementation plans that demonstrate that water quality standards can ultimately be met.

While some communities have started to restore or rehabilitate these streams in recent years, their efforts have yielded only modest improvements in water quality and biological indicators. In particular, no community has yet demonstrated that they can achieve water quality standards in an urban watershed that exceeds 25% IC. Many communities are deeply concerned that non-supporting streams may never achieve water quality standards, despite massive investments in watershed restoration. The ICM suggests that water quality standards may need to be sharply revised for streams with more than 25% IC, if they are ever to come into attainment. While states have authority to create more achievable standards for non-supporting streams within the regulatory framework of the Clean Water Act (Swietlik, 2001), no state has yet exercised this authority. At this time, we are not aware of any water quality standards that are based on the ICM or similar urban stream classification techniques.

Two political perceptions largely explain why states are so reticent about revising water quality standards. The first is a concern that they will run afoul of anti-degradation provisions within the Clean Water Act or be accused of "backsliding" by the environmental community. The second concern relates to the demographics of watershed organizations across the country. According to recent surveys, slightly more than half of all watershed organizations represent moderately to highly developed watersheds (CWP, 2001a). These urban watershed organizations often have a keen interest in keeping the existing regulatory structure intact, since it is perceived to be the only lever to motivate municipalities to implement restoration efforts in non-supporting streams.

However, revised water quality standards are urgently needed to support smart growth efforts. A key premise of smart growth is that it is more desirable to locate new development within a non-supporting subwatershed rather than a sensitive or impacted one (i.e., concentrating density and IC within an existing subwatershed helps prevent sprawl from encroaching on a less developed one). Yet while smart growth is desirable on a regional basis, it will usually contribute to already serious problems in non-supporting watersheds, which makes it even more difficult to meet water quality standards.

This creates a tough choice for regulators: if they adopt stringent development criteria for non-supporting watersheds, their added costs can quickly become a powerful barrier to desired redevelopment. If, on the other hand, they relax or waive environmental criteria, they contribute to the further degradation of the watershed. To address this problem, the Center has developed a "smart watersheds" program to ensure that any localized degradation caused by development within a non-supporting subwatershed is more than compensated for by improvements in stream quality achieved through municipal restoration efforts (CWP, in press). Specifically, the smart watersheds program includes 17 public sector programs to treat stormwater runoff, restore urban stream corridors and reduce pollution discharges in highly urban watersheds. It is hoped that communities that adopt and implement smart watershed programs will be given greater flexibility to meet state and federal water quality regulations and standards within non-supporting watersheds.

1.3.2 Use of the ICM for Urban Stream Classification

The ICM has proven to be a useful tool for classifying and managing the large inventory of streams that most communities possess. It is not unusual for a typical county to have several thousand miles of headwater streams within its political boundaries, and the ICM provides a unified framework to identify and manage these subwatersheds. In our watershed practice, we use the ICM to make an initial diagnosis rather than a final determination for stream classification. Where possible, we conduct rapid stream and subwatershed assessments as a final check for an individual stream classification, particularly if it borders between the sensitive and impacted category. As noted earlier, the statistical variation in the IC/stream quality indicator makes it difficult to distinguish between a stream with 9% versus 11% IC. Some of the key criteria we use to make a final stream classification are provided in Table 6.

1.3.3 Role of the ICM in Small Watershed Planning

The ICM has also proven to be an extremely important tool for watershed planning, since it can rapidly project how streams will change in response to future land use. We routinely estimate existing and future IC in our watershed planning practice and find that it is an excellent indicator of change for subwatersheds in the zero to 30% IC range. In particular, the ICM often forces watershed planners to directly confront land use planning and land conservation issues early in the planning process.

On the other hand, we often find that the ICM has limited planning value when subwatersheds exceed 30% IC for two practical reasons. First, the ICM does not differentiate stream conditions within this very large span of IC (i.e., there is no difference in the stream quality prediction for a subwatershed that has 39.6% IC versus one that has 58.4% IC). Second, the key management question for non-supporting watersheds is whether or not they are potentially restorable. More detailed analysis and field investigations are needed to determine, in each subwatershed, the answer to this question. While a knowledge of IC is often used in these feasibility assessments, it is but one of many factors that needs to be considered.

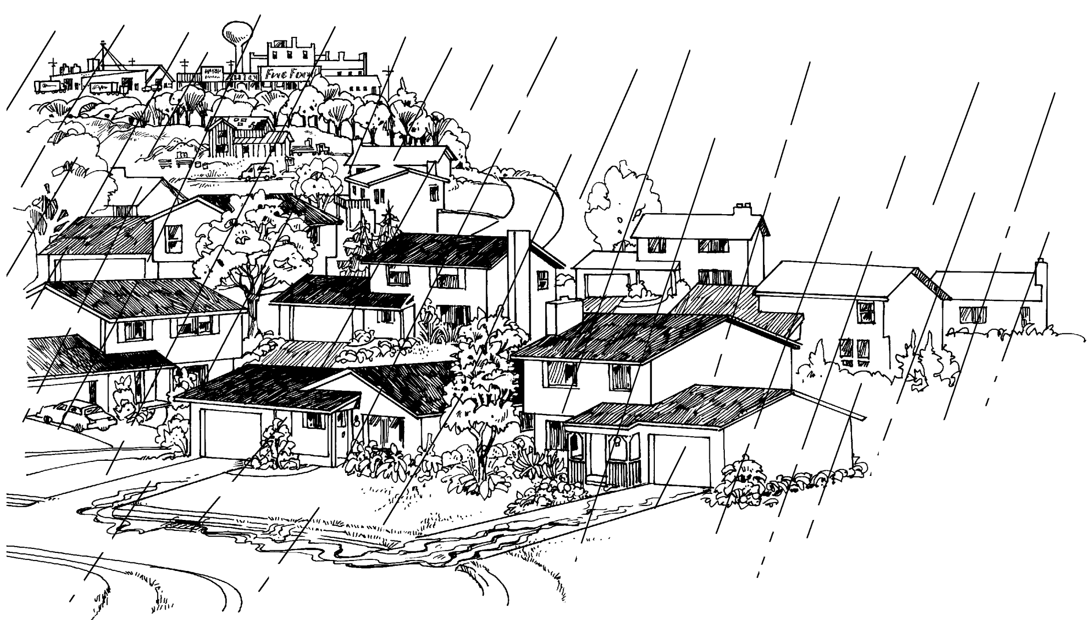
Lastly, we have come to recognize several practical factors when applying the ICM for small watershed planning. These include thoughtful delineation of subwatershed boundaries, the proper accounting of a direct drainage area in larger watersheds, and the critical need for the most recent IC data. More guidance on these factors can be found in Zielinski (2001).

Impervious cover is not a perfect indicator of existing stream quality. A number of stream and subwatershed criteria should be evaluated in the field before a final classification decision is made, particularly when the stream is on the borderline between two classifications. We routinely look at the stream and subwatershed criteria to decide whether a borderline stream should be classified as sensitive or impacted. Table 6 reviews these additional criteria.

Table 6: Additional Considerations for Urban Stream Classification
Stream Criteria
Reported presence of rare, threatened or endangered species in the aquatic community (e.g., freshwater mussels, fish, crayfish or amphibians)
Confirmed spawning of cold-water fish species (e.g., trout)
Fair/good, good, or good to excellent macro invertebrate scores
More than 65% of EPT species present in macro-invertebrate surveys
No barriers impede movement of fish between the subwatershed and downstream receiving waters
Stream channels show little evidence of ditching, enclosure, tile drainage or channelization
Water quality monitoring indicates no standards violations during dry weather
Stream and flood plain remain connected and regularly interact
Stream drains to a downstream surface water supply
Stream channels are generally stable, as determined by the Rosgen level analysis
Stream habitat scores are rated at least fair to good
Subwatershed Criteria
Contains terrestrial species that are documented as rare, threatened and endangered
Wetlands, flood plains and/or beaver complexes make up more than 10% of subwatershed area
Inventoried conservation areas comprise more than 10% of subwatershed area
More than 50% of the riparian forest corridor has forest cover and is either publicly owned or regulated
Large contiguous forest tracts remain in the subwatershed (more than 40% in forest cover)
Significant fraction of subwatershed is in public ownership and management
Subwatershed connected to the watershed through a wide corridor
Farming, ranching and livestock operations in the subwatershed utilize best management practices
Prior development in the subwatershed has utilized stormwater treatment practices

1.4 Summary

The remainder of this report presents greater detail on the individual research studies that bear on the ICM. Chapter 2 profiles research on hydrologic indicators in urban streams, while Chapter 3 summarizes the status of current research on the impact of urbanization on physical habitat indicators. Chapter 4 presents a comprehensive review of the impact of urbanization on ten major stormwater pollutants. Finally, Chapter 5 reviews the growing body of research on the link between IC and biological indicators within urban streams and wetlands.



Chapter 2: Hydrologic Impacts of Impervious Cover

The natural hydrology of streams is fundamentally changed by increased watershed development. This chapter reviews the impacts of watershed development on selected indicators of stream hydrology.

This chapter is organized as follows:

2.1 Introduction
2.2 Increased Runoff Volume
2.3 Increased Peak Discharge Rates
2.4 Increased Bankfull Flow
2.5 Decreased Baseflow
2.6 Conclusions

2.1 Introduction

Fundamental changes in urban stream hydrology occur as a result of three changes in the urban landscape that accompany land development. First, large areas of the watershed are paved, rendering them impervious. Second, soils are compacted during construction, which significantly reduces their infiltration capabilities. Lastly, urban stormwater drainage systems are installed that increase the efficiency with which runoff is delivered to the stream (i.e., curbs and gutters, and storm drain pipes). Consequently, a greater fraction of annual rainfall is converted to surface runoff, runoff occurs more quickly, and peak flows become larger. Additionally, dry weather flow in streams may actually decrease because less groundwater recharge is available. Figure 6 illustrates the change in hydrology due to increased urban runoff as compared to pre-development conditions.

Research has demonstrated that the effect of watershed urbanization on peak discharge is more marked for smaller storm events. In particular, the bankfull, or channel forming flow, is increased in magnitude, frequency and duration. Increased bankfull flows have strong ramifications for sediment transport and channel enlargement. All of these changes in the natural water balance have impacts on the physical structure of streams, and ultimately affect water quality and biological diversity.

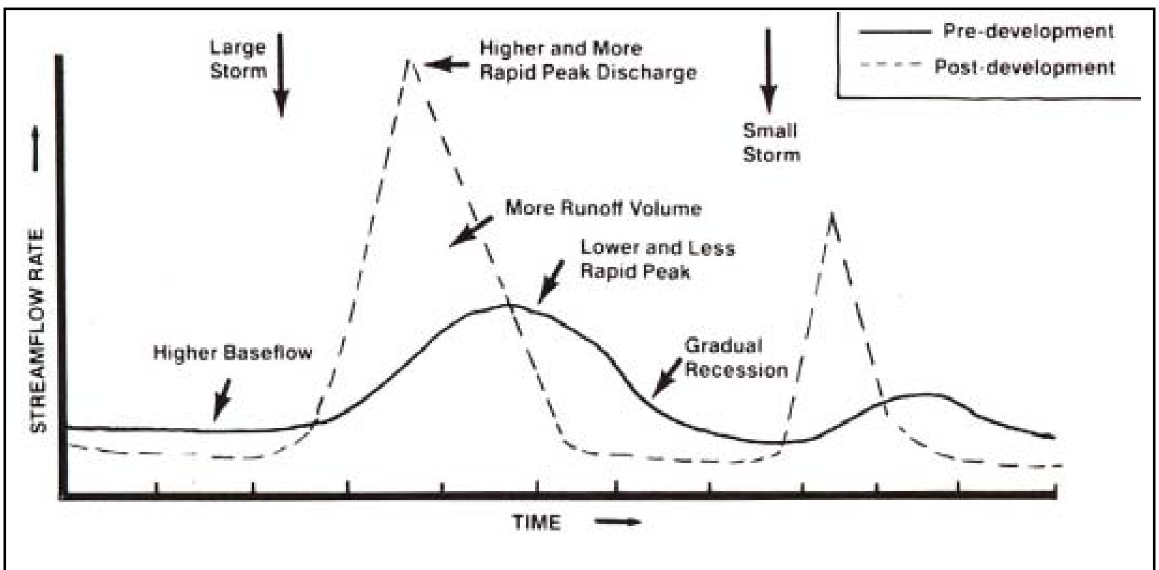


Figure 6: Altered Hydrograph in Response to Urbanization (Schueler, 1987)

The relationship between watershed IC and stream hydrology is widely accepted, and has been incorporated into many hydrologic engineering models over the past three decades. Several articles provide a good summary of these (Bicknell *et al.*, 1993; Hirsch *et al.*, 1990; HEC, 1977; Huber and Dickinson, 1988; McCuen and Moglen, 1988; Overton and Meadows, 1976; Pitt and Voorhees, 1989; Schueler, 1987; USDA, 1992; 1986).

The primary impacts of watershed development on stream hydrology are as follows:

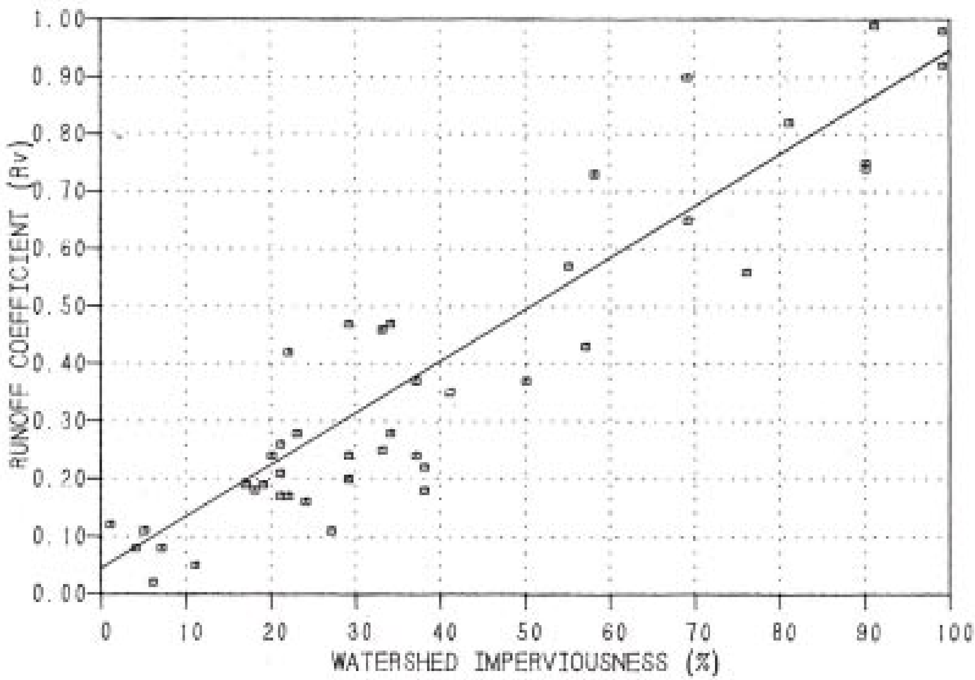
- Increased runoff volume
- Increased peak discharge rates
- Increased magnitude, frequency, and duration of bankfull flows
- Diminished baseflow

2.2 Increased Runoff Volume

Impervious cover and other urban land use alterations, such as soil compaction and storm drain construction, alter infiltration rates and increase runoff velocities and the efficiency with which water is delivered to streams. This decrease in infiltration and basin lag time can significantly increase runoff volumes. Table 7 reviews research on the impact of IC on runoff volume in urban streams. Schueler (1987) demonstrated that runoff values are directly related to subwatershed IC (Figure 7). Runoff data was derived from 44 small catchment areas across the country for EPA's Nationwide Urban Runoff Program.

Table 8 illustrates the difference in runoff volume between a meadow and a parking lot, as compiled from engineering models. The parking lot produces more than 15 times more runoff than a meadow for the same storm event.

Urban soils are also profoundly modified during the construction process. The compaction of urban soils and the removal of topsoil can decrease the infiltration capacity, causing increases in runoff volumes (Schueler, 2000). Bulk density is often used to measure soil compaction, and Table 9 illustrates how bulk density increases in many urban land uses.



Note: 44 small urban catchments monitored during the national NURP study

Figure 7: Runoff Coefficient vs. IC (Schueler, 1987)

Table 7: Research Review of Increased Runoff Volume and Peak Discharge in Urban Streams		
Reference	**Key Finding**	**Location**
	Increased Runoff Volume	
Schueler, 1987	Runoff coefficients were found to be strongly correlated with IC at 44 sites nationwide.	U.S.
Neller, 1988	Urban watershed produced more than seven times as much runoff as a similar rural watershed. Average time to produce runoff was reduced by 63% in urban watersheds compared to rural watersheds.	Australia
	Increased Peak Discharge	
Hollis, 1975	Review of data from several studies showed that floods with a return period of a year or longer are not affected by a 5% watershed IC; small floods may be increased 10 times by urbanization; flood with a return period of 100 years may be doubled in size by a 30% watershed IC.	N/A
Leopold, 1968	Data from seven nationwide studies showed that 20% IC can cause the mean annual flood to double.	U.S.
Neller, 1988	Average peak discharge from urban watersheds was 3.5 times higher than peak runoff from rural watersheds.	Australia
Doll *et al.*, 2000	Peak discharge was greater for 18 urban streams versus 11 rural Piedmont streams.	NC
Sauer *et al.*, 1983	Estimates of flood discharge for various recurrence intervals showed that less than 50% watershed IC can result in a doubling of the 2-year, 10-year, and 100-year floods.	U.S.
Leopold, 1994	Watershed development over a 29-year period caused the peak discharge of the 10-year storm to more than double.	MD
Kibler *et al.*, 1981	Rainfall/runoff model for two watersheds showed that an increase in IC caused a significant increase in mean annual flood.	PA
Konrad and Booth, 2002	Evaluated streamflow data at 11 streams and found that the fraction of annual mean discharges was exceeded and maximum annual instantaneous discharges were related to watershed development and road density for moderately and highly developed watersheds.	WA

Table 8: Hydrologic Differences Between a Parking Lot and a Meadow (Schueler, 1994a)		
Hydrologic or Water Quality Parameter	**Parking Lot**	**Meadow**
Runoff Coefficient	0.95	0.06
Time of Concentration (minutes)	4.8	14.4
Peak Discharge, two-year, 24-hour storm (cfs)	4.3	0.4
Peak Discharge Rate, 100-year storm (cfs)	12.6	3.1
Runoff Volume from one-inch storm (cu. ft)	3,450	218
Runoff Velocity @ two-year storm (ft/sec)	8	1.8
Key Assumptions: *2-yr, 24-hr storm = 3.1 in; 100-yr storm = 8.9 in.* *Parking Lot: 100% imperviousness; 3% slope; 200ft flow length; hydraulic radius =.03; concrete channel; suburban Washington C values* *Meadow: 1% impervious; 3% slope; 200 ft flow length; good vegetative condition; B soils; earthen channel* *Source: Schueler, 1994a*		

Table 9: Comparison of Bulk Density for Undisturbed Soils and Common Urban Conditions (Schueler, 2000)			
Undisturbed Soil Type or Urban Condition	**Surface Bulk Density (grams/cubic centimeter)**	**Urban Condition**	**Surface Bulk Density (grams/cubic centimeter)**
Peat	0.2 to 0.3	Urban Lawns	1.5 to 1.9
Compost	1.0	Crushed Rock Parking Lot	1.5 to 1.9
Sandy Soils	1.1 to 1.3	Urban Fill Soils	1.8 to 2.0
Silty Sands	1.4	Athletic Fields	1.8 to 2.0
Silt	1.3 to 1.4	Rights-of-Way and Building Pads (85%)	1.5 to 1.8
Silt Loams	1.2 to 1.5	Rights-of-Way and Building Pads (95%)	1.6 to 2.1
Organic Silts/Clays	1.0 to 1.2	Concrete Pavement	2.2
Glacial Till	1.6 to 2.0	Rock	2.65

2.3 Increased Peak Discharge Rate

Watershed development has a strong influence on the magnitude and frequency of flooding in urban streams. Peak discharge rates are often used to define flooding risk. Doll *et al.* (2000) compared 18 urban streams with 11 rural streams in the North Carolina Piedmont and found that unit area peak discharge was always greater in urban streams (Figure 8). Data from Seneca Creek, Maryland also suggest a similar increase in peak discharge. The watershed experienced significant growth during the 1950s and 1960s. Comparison of pre- and post-development gage records suggests that the peak 10-year flow event more than doubled over that time (Leopold, 1994).

Hollis (1975) reviewed numerous studies on the effects of urbanization on floods of different recurrence intervals and found that the effect of urbanization diminishes when flood recurrence gets longer (i.e., 50 and 100 years). Figure 9 shows the effect on flood magnitude in urban watersheds with 30% IC, and shows the one-year peak discharge rate increasing by a factor of 10, compared to an undeveloped watershed. In contrast, floods with a 100-year recurrence interval only double in size under the same watershed conditions.

Sauer *et al.* (1983) evaluated the magnitude of flooding in urban watersheds throughout the United States. An equation was developed for estimating discharge for floods of two-year, 10-year, and 100-year recurrence intervals. The equations used IC to account for increased runoff volume and a basin development factor to account for sewers, curbs and gutters, channel improvements and drainage development. Sauer noted that IC is not the dominant factor in determining peak discharge rates for extreme floods because these storm events saturate the soils of undeveloped watersheds and produce high peak discharge rates. Sauer found that watersheds with 50% IC can increase peak discharge for the two-year flood by a factor of four, the 10-year flood by a factor of three, and the 100-year flood by a factor of 2.5, depending on the basin development factor (Figure 10).

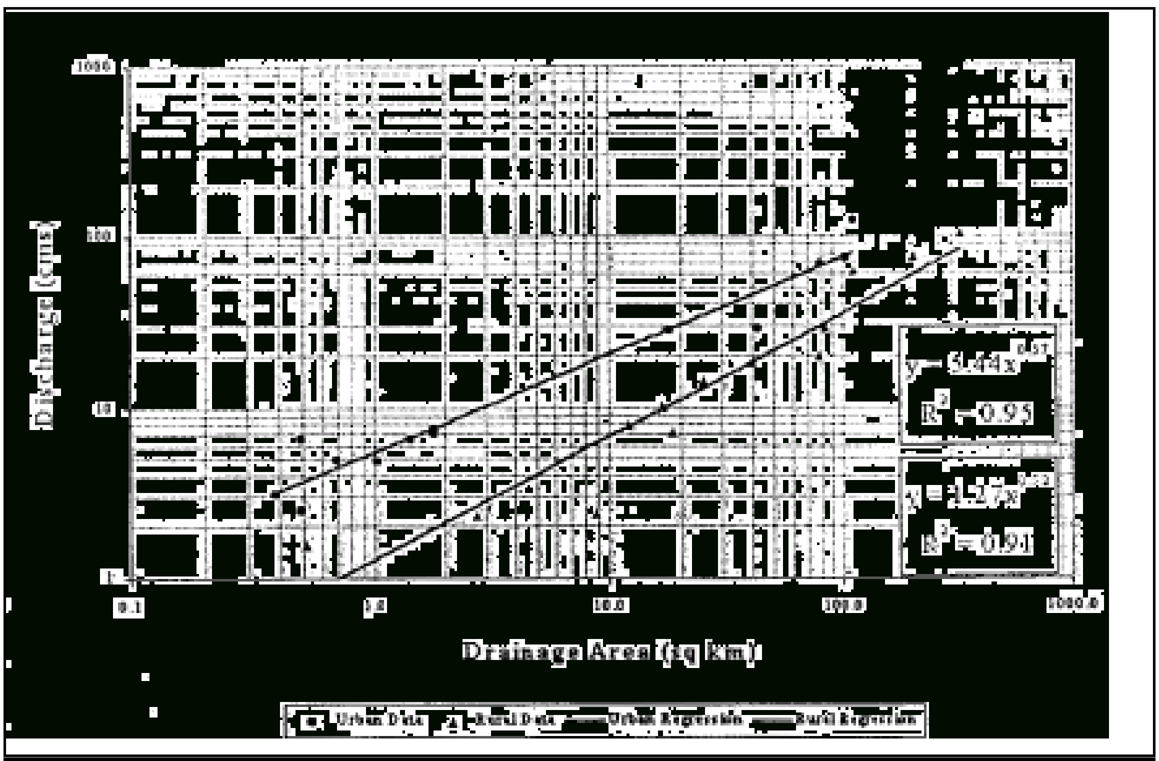


Figure 8: Peak Discharge for Urban and Rural Streams in North Carolina (Doll *et al.*, 2000)

2.4 Increased Bankfull Flow

Urbanization also increases the frequency and duration of peak discharge associated with smaller flood events (i.e., one- to two-year return storms). In terms of stream channel morphology, these more frequent bankfull flows are actually much more important than large flood events in forming the channel. In fact, Hollis (1975) demonstrated that urbanization increased the frequency and magnitude of bankfull flow events to a greater degree than the larger flood events.

An example of the increase in bankfull flow in arid regions is presented by the U.S. Geological Survey (1996), which compared the peak discharge rate from two-year storm events before and after watersheds urbanized in Parris Valley, California. Over an approximately 20-year period, watershed IC increased by 13.5%, which caused the two-year peak flow to more than double. Table 10 reviews other research studies on the relationship between watershed IC and bankfull flows in urban streams.

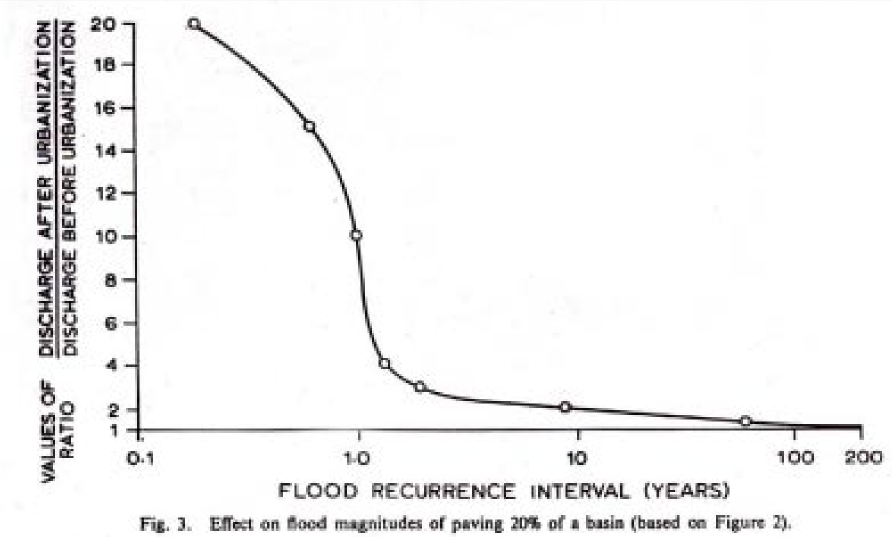


Figure 9: Effect on Flood Magnitudes of 30% Basin IC (Hollis, 1975)

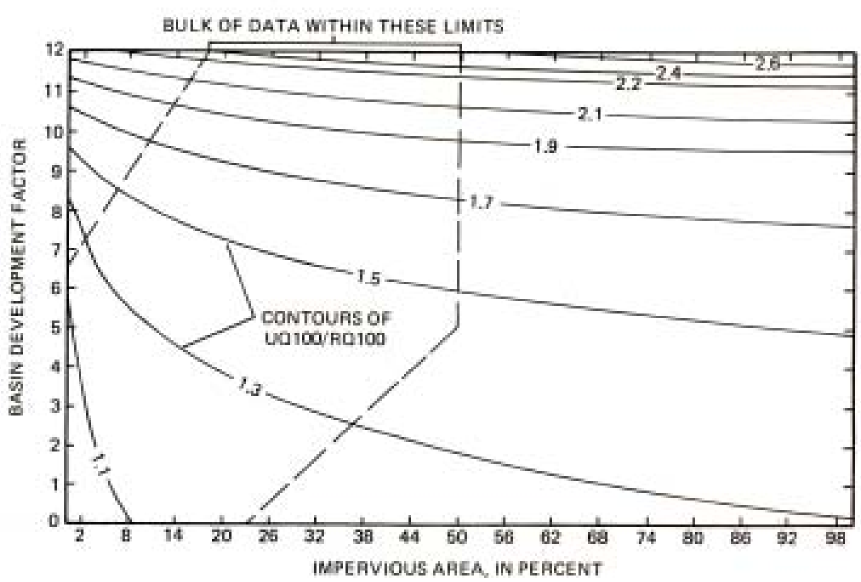


Figure 10: Relationship of Urban/Rural 100-Year Peak Flow Ratio to Basin Development Factor and IC (Sauer *et al.*, 1983)

Table 10: Research Review of Increased Bankfull Discharge in Urban Streams		
Reference	**Key Finding**	**Location**
Booth and Reinelt, 1993	Using a simulation model and hydrologic data from four watersheds, it was estimated that more than 10% watershed IC may cause discharge from the two-year storm under current conditions to equal or exceed discharge from the 10-year storm under forested conditions.	WA
Fongers and Fulcher, 2001	Bankfull flow of 1200 cfs was exceeded more frequently over time with urbanization, and exceedence was three times as frequent from 1930s to 1990s.	MI
USGS, 1996	Over a 20-year period, IC increased 13.5%, and the two-year peak flow more than doubled in a semi-arid watershed.	CA
Henshaw and Booth, 2000	Two of three watersheds in the Puget Sound lowlands showed increasing flashiness over 50 years with urbanization.	WA
Leopold, 1968	Using hydrologic data from a nine-year period for North Branch Brandywine Creek, it was estimated that for a 50% IC watershed, bankfull frequency would be increased fourfold.	PA
Leopold, 1994	Bankfull frequency increased two to seven times after urbanization in Watts Branch.	MD
MacRae, 1996	For a site downstream of a stormwater pond in Markham, Ontario hours of exceedence of bankfull flows increased by 4.2 times after the watershed urbanized (34% IC)	Ontario

Leopold (1968) evaluated data from seven nationwide studies and extrapolated this data to illustrate the increase in bankfull flows due to urbanization. Figure 11 summarizes the relationship between bankfull flows over a range of watershed IC. For example, watersheds that have 20% IC increase the number of flows equal to or greater than bankfull flow by a factor of two. Leopold (1994) also observed a dramatic increase in the frequency of the bankfull event in Watts Branch, an urban subwatershed in Rockville, Maryland. This watershed experienced significant urban development during the 1950s and 1960s. Leopold compared gage records and found that the bankfull storm event frequency increased from two to seven times per year from 1958 to 1987.

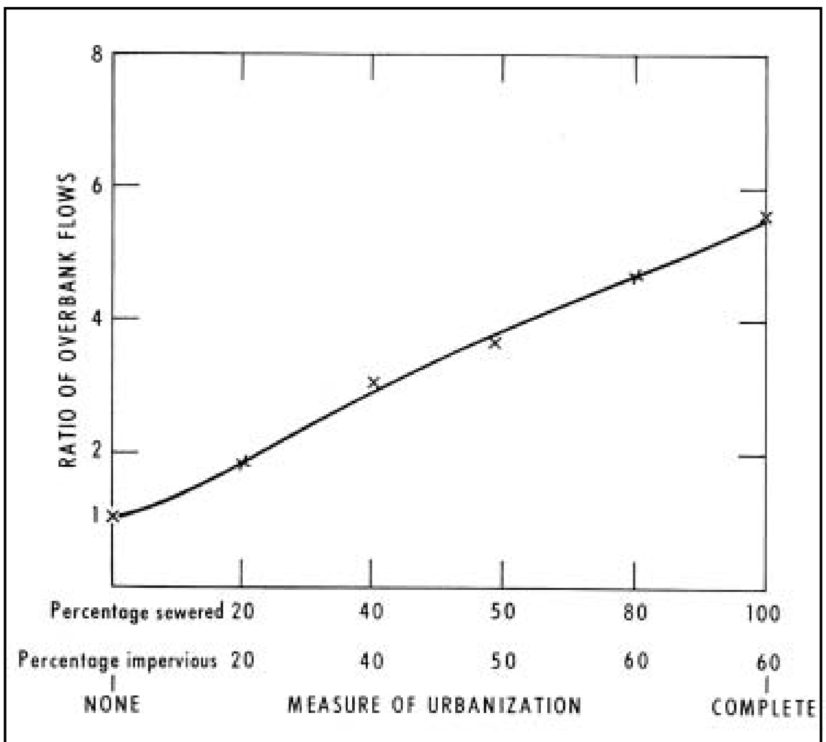


Figure 11: Increase in Bankfull Flows Due to Urbanization (Leopold, 1968)

More recent data on bankfull flow frequency was reported for the Rouge River near Detroit, Michigan by Fongers and Fulcher (2001). They noted that channel-forming flow (1200 cfs) was exceeded more frequently as urbanization increased in the watershed and had become three times more frequent between 1930 and 1990 (Figure 12).

McCuen and Moglen (1988) have documented the increase in duration of bankfull flows in response to urbanization using hydrology models. MacRae (1996), monitored a stream in Markham, Ontario downstream of a stormwater pond and found that the hours of

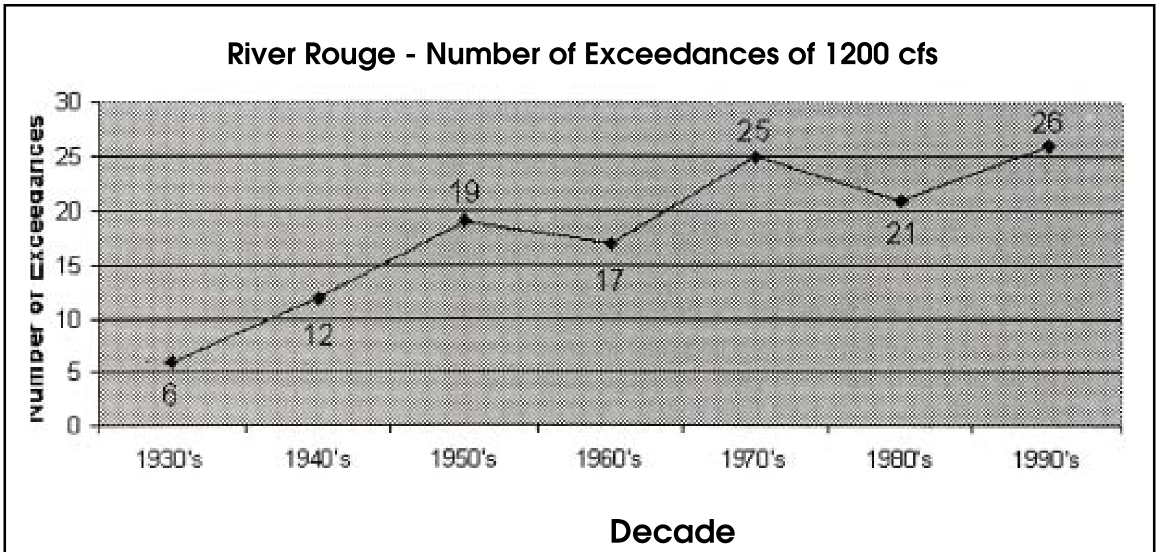


Figure 12: Increase in Number of Exceedences of Bankfull Flow Over Time With Urbanization in the Rouge River, MT (Fongers and Fulcher, 2001)

exceedence of bankfull flows increased by a factor of 4.2 once watershed IC exceeded 30%. Modeling for seven streams also downstream of stormwater ponds in Surrey, British Columbia also indicated an increase in bankfull flooding in response to watershed development (MacRae, 1996).

Watershed IC also increases the "flashiness" of stream hydrographs. Flashiness is defined here as the percent of daily flows each year that exceeds the mean annual flow. Henshaw and Booth (2000) evaluated seven urbanized watersheds in the Puget Sound lowland streams and tracked changes in flashiness over 50 years (Figure 13). The most urbanized watersheds experienced flashy discharges. Henshaw and Booth concluded that increased runoff in urban watersheds leads to higher but shorter-duration peak discharges.

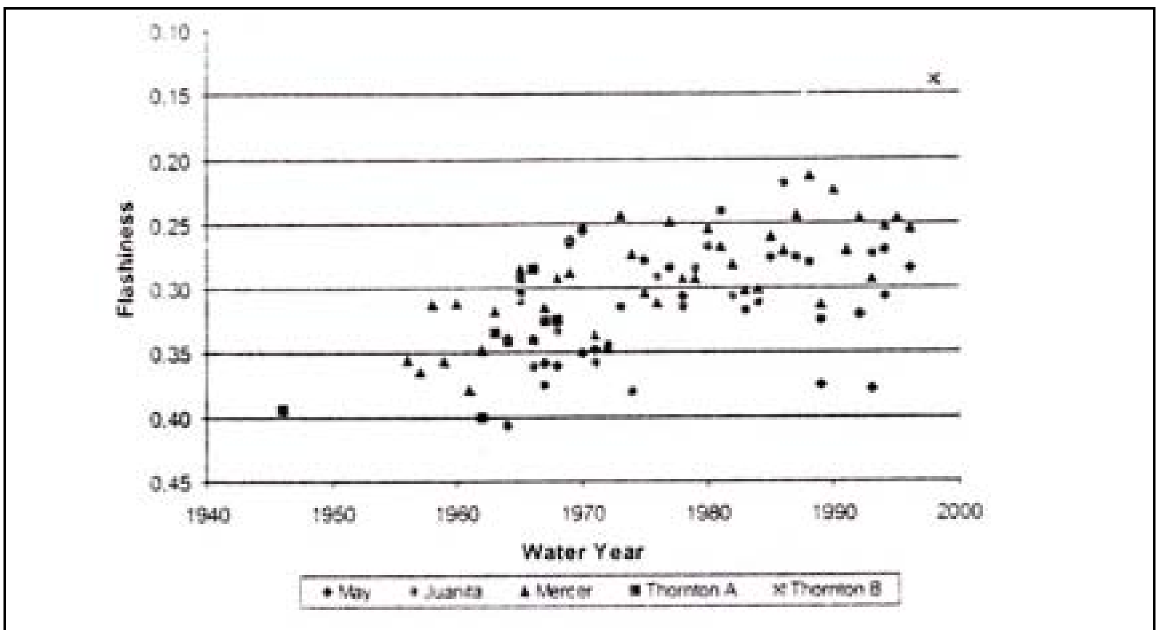


Figure 13: Percent of Gage Reading Above Mean Annual Flow for Puget Sound Lowland Streams (Henshaw and Booth, 2000)

2.5 Decreased Baseflow

As IC increases in a watershed, less groundwater infiltration is expected, which can potentially decrease stream flow during dry periods, (i.e. baseflow). Several East Coast studies provide support for a decrease in baseflow as a result of watershed development. Table 11 reviews eight research studies on baseflow in urban streams.

Klein (1979) measured baseflow in 27 small watersheds in the Maryland Piedmont and reported an inverse relationship between IC and baseflow (Figure 14). Spinello and Simmons (1992) demonstrated that baseflow in two urban Long Island streams declined seasonally as a result of urbanization (Figure 15). Saravanapavan (2002) also found that percentage of baseflow decreased in direct proportion to percent IC for 13 subwatersheds of the Shawsheen River watershed in Massachusetts (Figure 16).

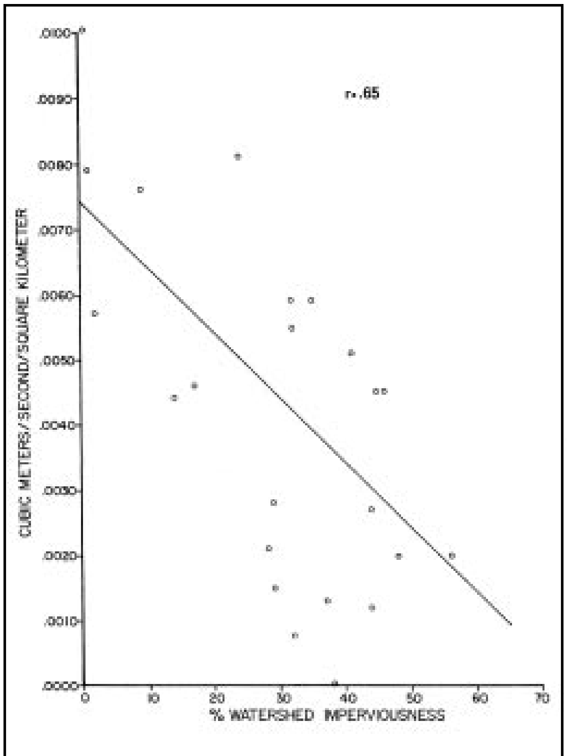
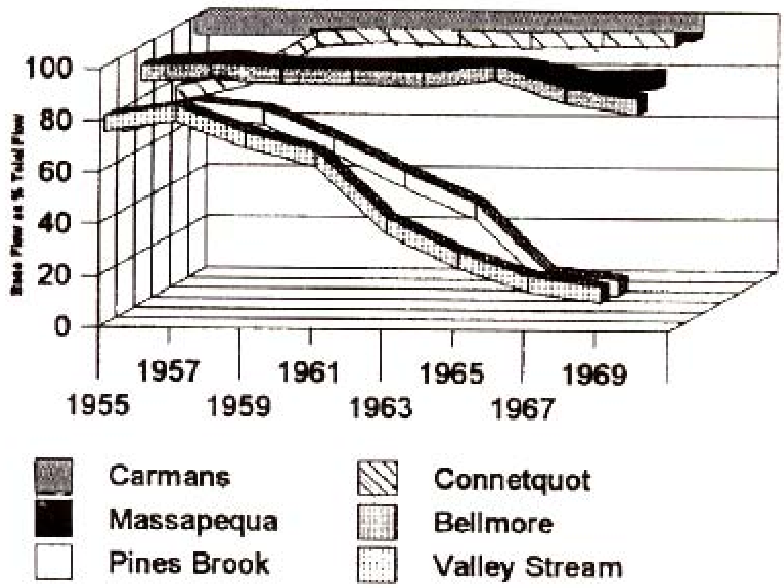


Figure 14: Relationship Between Baseflow and Watershed IC in the Streams on Maryland Piedmont (Klein, 1979)

Table 11: Research Review of Decreased Baseflow in Urban Streams		
Reference	**Key Finding**	**Location**
Finkenbine *et al.*, 2000	Summer base flow was uniformly low in 11 streams when IC reached 40% or greater.	Vancouver
Klein, 1979	Baseflow decreased as IC increased in Piedmont streams.	MD
Saravanapavan, 2002	Percentage of baseflow decreased linearly as IC increased for 13 subwatersheds of Shawsheen River watershed.	MA
Simmons and Reynolds, 1982	Dry weather flow dropped 20 to 85% after development in several urban watersheds on Long Island.	NY
Spinello and Simmons, 1992	Baseflow in two Long Island streams went dry as a result of urbanization.	NY
Konrad and Booth, 2002	No discernable trend over many decades in the annual seven day low flow discharge for 11 Washington streams.	WA
Wang *et al.*, 2001	Stream baseflow was negatively correlated with watershed IC in 47 small streams, with an apparent breakpoint at 8 to 12% IC.	WI
Evett *et al.*, 1994	No clear relationship between dry weather flow and urban and rural streams in 21 larger watersheds.	NC



Notes:

Carmans and Connetquot: Rural/suburban, watersheds are unsewered

Bellmore and Massapequa: Moderately urbanized, not sewered until 1989

Valley Stream and Pine Brook: Urbanized and densely populated watersheds, sewer systems completed in 1960s

1953-1964: Period of sanitary sewer construction in eastern Nassau County

1962-1966: Drought years

Figure 15: Baseflow Response to Urbanization in Long Island Streams (Spinello and Simmons, 1992)

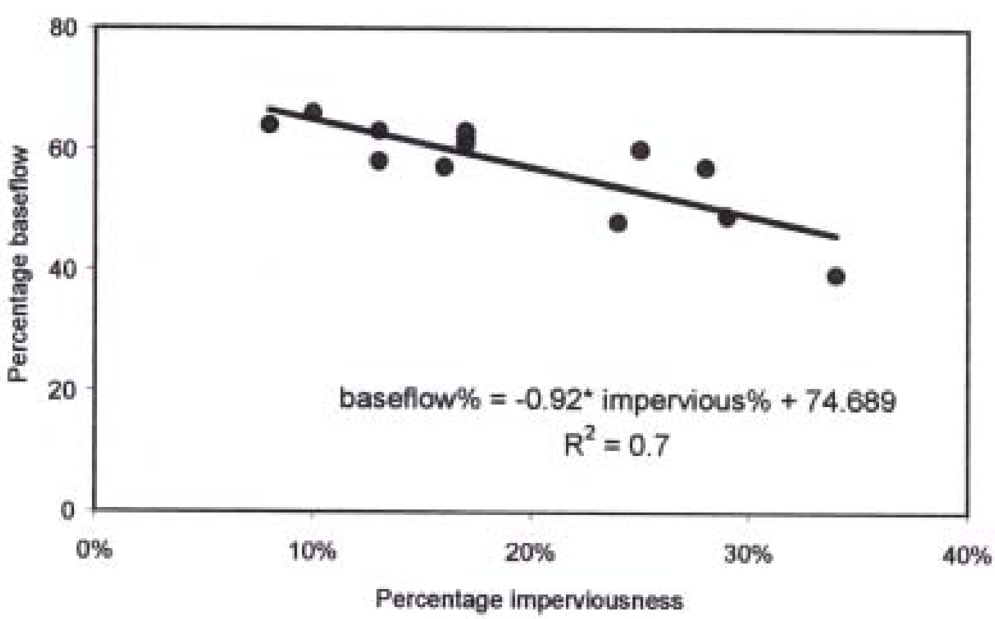


Figure 16: Relationship Between Percentage Baseflow and Percent IC in Massachusetts Streams (Saravanapan, 2002)

Finkebine *et al.* (2000) monitored summer baseflow in 11 streams near Vancouver, British Columbia and found that stream base flow was uniformly low due to decreased groundwater recharge in watersheds with more than 40% IC (Figure 17). Baseflow velocity also consistently decreased when IC increased (Figure 18). The study cautioned that other factors can affect stream baseflow, such as watershed geology and age of development.

Other studies, however, have not been able to establish a relationship between IC and declining baseflow. For example, a study in North Carolina could not conclusively determine that urbanization reduced baseflow in larger urban and suburban watersheds in that area (Evett *et al.*, 1994). In some cases, stream baseflow is supported by deeper aquifers or originate in areas outside the surface watershed boundary. In others, baseflow is augmented by leaking sewers, water pipes and irrigation return flows.

This appears to be particularly true in arid and semi-arid areas, where baseflow can actually increase in response to greater IC (Hollis, 1975). For instance, Crippen and Waananen (1969) found that Sharon Creek near San Francisco changed from an ephemeral stream into a perennial stream after urban development. Increased infiltration from lawn watering and return flow from sewage treatment plants are two common sources of augmented baseflows in these regions (Caraco, 2000a).

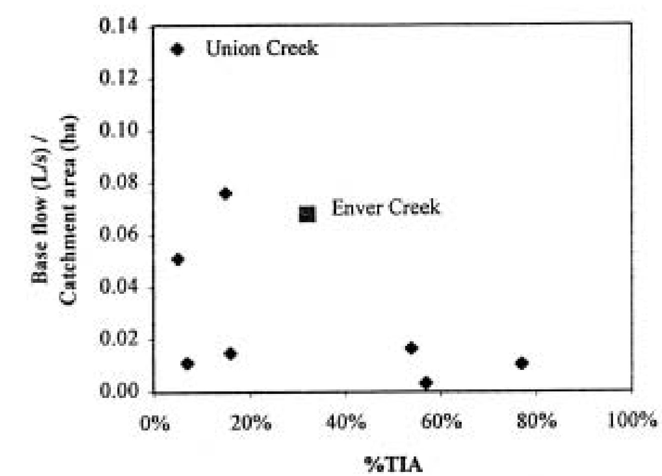


Figure 17: Effect of IC on Summer Baseflow in Vancouver Streams (Finkerbine *et al.*, 2000)

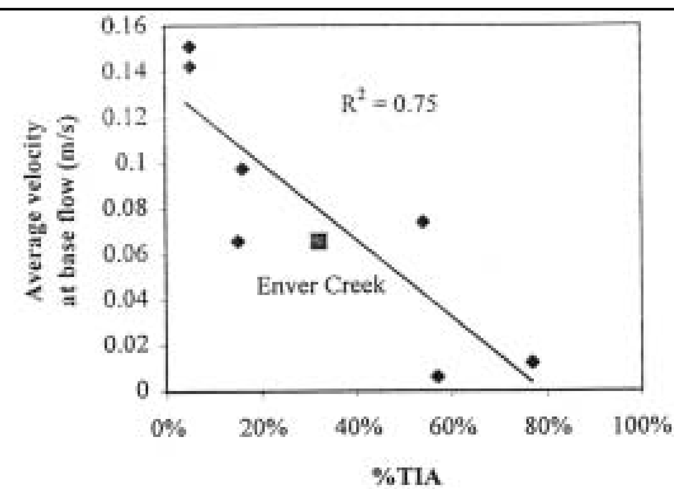


Figure 18: Effect of Watershed IC on Summer Stream Velocity in Vancouver Streams (Finkerbine *et al.*, 2000)

2.6 Conclusions

The changes in hydrology indicators caused by watershed urbanization include increased runoff volume; increased peak discharge; increased magnitude, frequency and duration of bankfull flows; flashier/less predictable flows; and decreased baseflow. Many studies support the direct relationship between IC and these indicators. However, at low levels of watershed IC, site-specific factors such as slope, soils, types of conveyance systems, age of development, and watershed dimensions often play a stronger role in determining a watershed's hydrologic response.

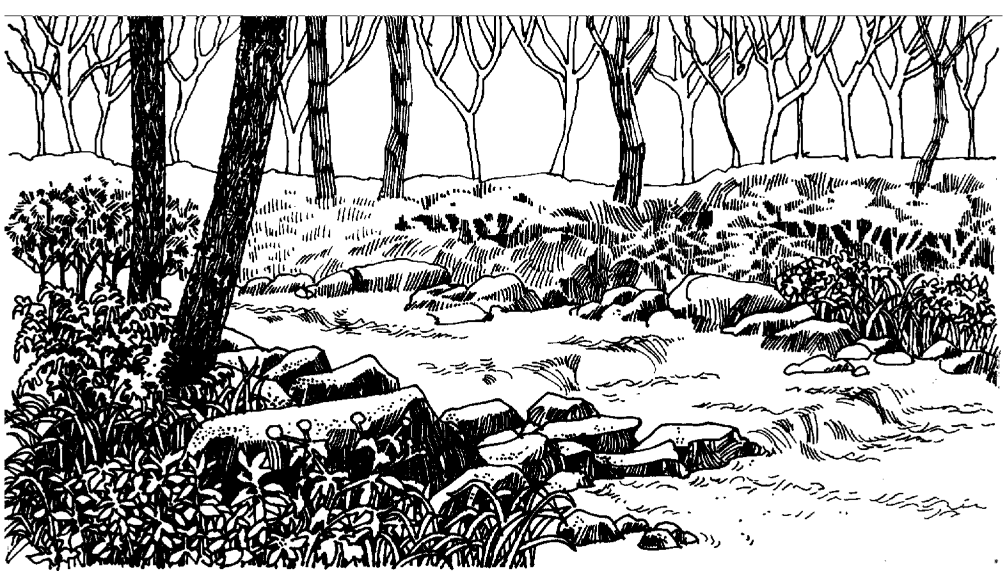
Overall, the following conclusions can be drawn from the relationship between watershed IC and hydrology indicators:

- Strong evidence exists for the direct relationship between watershed IC and increased stormwater runoff volume and peak discharge. These relationships are considered so strong that they have been incorporated into widely accepted engineering models.

- The relationship between IC and bankfull flow frequency has not been extensively documented, although abundant data exists for differences between urban and non-urban watersheds.

- The relationship between IC and declining stream flow is more ambiguous and appears to vary regionally in response to climate and geologic factors, as well as water and sewer infrastructure.

The changes in hydrology indicators caused by watershed urbanization directly influence physical and habitat characteristics of streams. The next chapter reviews how urban streams physically respond to the major changes to their hydrology.



Chapter 3: Physical Impacts of Impervious Cover

A growing body of scientific literature documents the physical changes that occur in streams undergoing watershed urbanization. This chapter discusses the impact of watershed development on various measures of physical habitat in urban stream channels and is organized as follows:

3.1 Difficulty in Measuring Habitat
3.2 Changes in Channel Geometry
3.3 Effect on Composite Indexes of Stream Habitat
3.4 Effect on Individual Elements of Stream Habitat
3.5 Increased Stream Warming
3.6 Alteration of Stream Channel Network
3.7 Conclusion

This chapter reviews the available evidence on stream habitat. We begin by looking at geomorphological research that has examined how the geometry of streams changes in response to altered urban hydrology. The typical response is an enlargement of the cross-sectional area of the stream channel through a process of channel incision, widening, or a combination of both. This process triggers an increase in bank and/or bed erosion that increases sediment transport from the stream, possibly for several decades or more.

Next, we examine the handful of studies that have evaluated the relationship between watershed development and composite indicators of stream habitat (such as the habitat Rapid Bioassessment Protocol, or RBP). In the fourth section, we examine the dozen studies that have evaluated how individual habitat elements respond to watershed development. These studies show a consistent picture. Generally, streams with low levels of IC have stable banks, contain considerable large woody debris (LWD) and possess complex habitat structure. As watershed IC increases, however, urban streambanks become increasingly unstable, streams lose LWD, and they develop a more simple and uniform habitat structure. This is typified by reduced pool depths, loss of pool and riffle sequences, reduced channel roughness and less channel sinuosity.

Water temperature is often regarded as a key habitat element, and the fifth section describes the stream warming effect observed in urban streams in six studies. The last section looks at the effect of watershed development on the stream channel network as a whole, in regard to headwater stream loss and the creation of fish barriers.

3.1 Difficulty in Measuring Habitat

The physical transformation of urban streams is perhaps the most conspicuous impact of watershed development. These dramatic physical changes are easily documented in sequences of stream photos with progressively greater watershed IC (see Figure 19). Indeed, the network of headwater stream channels generally disappears when watershed IC exceeds 60% (CWP).

3.1.1 The Habitat Problem

It is interesting to note that while the physical impacts of urbanization on streams are widely accepted, they have rarely been documented by the research community. As a consequence, no predictive models exist to quantify how physical indicators of stream habitat will decline in response to watershed IC, despite the fact that most would agree that some kind of decline is expected (see Table 12).

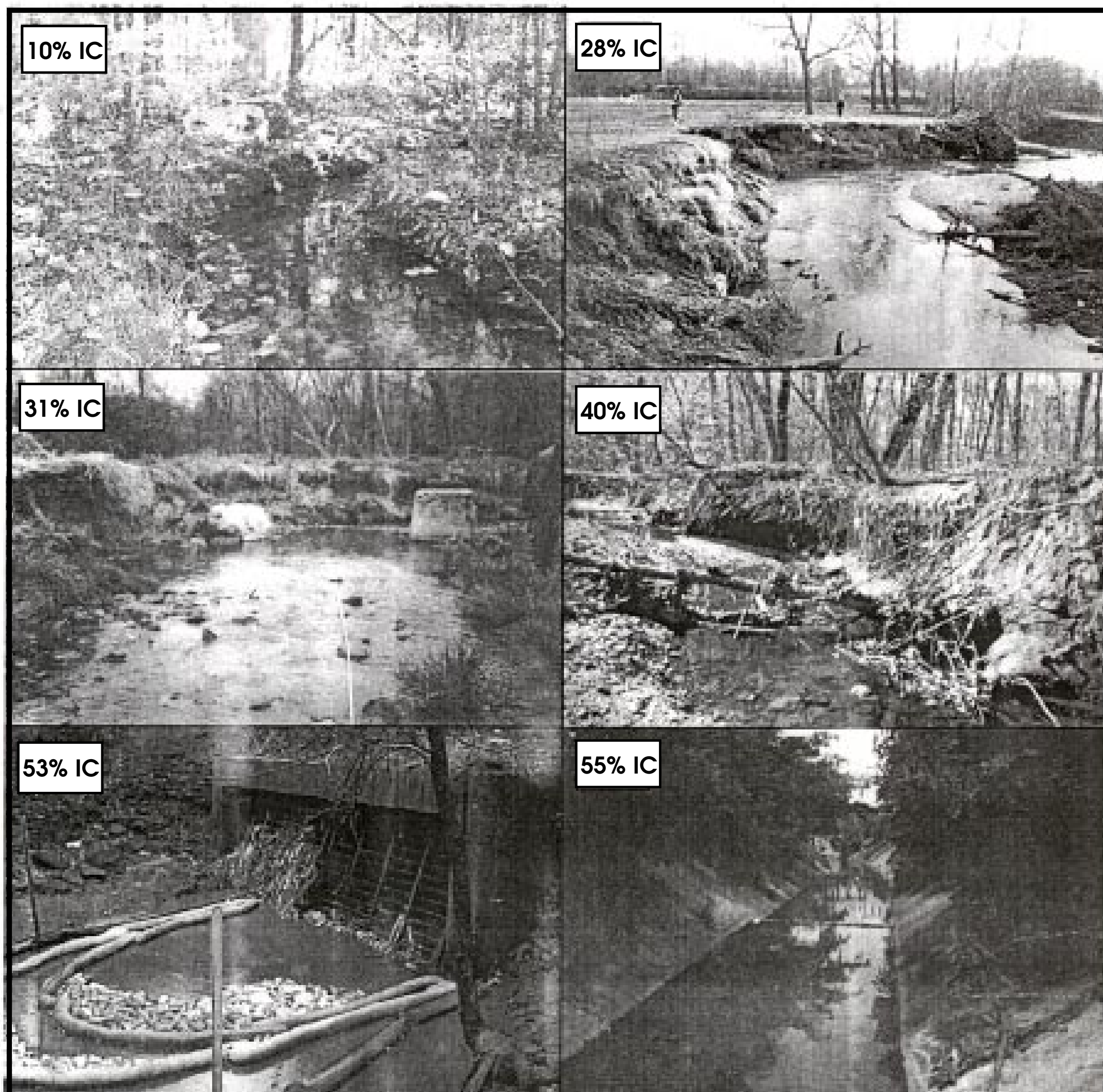


Figure 19: Urban Stream Channels with Progressively Greater IC

The main reason for this gap is that "habitat" is extremely hard to define, and even more difficult to measure in the field. Most indices of physical habitat involve a visual and qualitative assessment of 10 or more individual habitat elements that are perceived by fishery and stream biologists to contribute to quality stream habitat. Since these indices include many different habitat elements, each of which is given equal weight, they have not been very useful in discriminating watershed effects (Wang *et al.*, 2001).

Researchers have had greater success in relating individual habitat elements to watershed conditions, such as large woody debris (LWD), embeddedness, or bank stability. Even so, direct testing has been limited, partly because individual habitat elements are hard to measure and are notoriously variable in both space and time. Consider bank stability for a moment. It would be quite surprising to see a highly urban stream that did not have unstable banks. Yet, the hard question is exactly how would bank instability be quantitatively measured? Where would it be measured — at a point, a cross-section, along a reach, on the left bank or the right?

Geomorphologists stress that no two stream reaches are exactly alike, due to differences in gradient, bed material, sediment transport, hydrology, watershed history and many other factors. Consequently, it is difficult to make controlled comparisons among different streams. Indeed, geomorphic theory stresses that individual stream reaches respond in a highly dynamic way to changes in watershed hydrology and sediment transport, and can take several decades to fully adjust to a new equilibrium.

Table 12: Physical Impacts of Urbanization on Streams

Specific Impacts
Sediment transport modified
Channel enlargement
Channel incision
Stream embeddedness
Loss of large woody debris
Changes in pool/riffle structure
Loss of riparian cover
Reduced channel sinuosity
Warmer in-stream temperatures
Loss of cold water species and diversity
Channel hardening
Fish blockages
Loss of 1st and 2nd order streams through storm drain enclosure

Returning to our example of defining bank stability, how might our measure of bank instability change over time as its watershed gradually urbanizes, is built out, and possibly reaches a new equilibrium over several decades? It is not very surprising that the effect of watershed development on stream habitat is widely observed, yet rarely measured.

3.2 Changes in Stream Geometry

As noted in the last chapter, urbanization causes an increase in the frequency and duration of bankfull and sub-bankfull flow events in streams. These flow events perform more "effective work" on the stream channel, as defined by Leopold (1994). The net effect is that an urban stream channel is exposed to more shear stress above the critical threshold needed to move bank and bed sediments (Figure 20). This usually triggers a cycle of active bank erosion and greater sediment transport in urban streams. As a consequence, the stream channel adjusts by expanding its cross-sectional area, in order to effectively accommodate greater flows and sediment supply. The stream channel can expand by incision, widening, or both. Incision refers to stream down-cutting through the streambed, whereas widening refers to lateral erosion of the stream bank and its flood plain (Allen and Narramore, 1985; Booth, 1990; Morisawa and LaFlure, 1979).

3.2.1 Channel Enlargement

A handful of research studies have specifically examined the relationship between watershed development and stream channel enlargement (Table 13). These studies indicate that stream cross-sectional areas can enlarge by as much as two to eight times in response to urbanization, although the process is complex and may take several decades to complete (Pizzuto *et al.*, 2000; Caraco, 2000b; Hammer, 1972). An example of channel enlargement is provided in Figure 21, which shows how a stream cross-section in Watts Branch near Rockville, Maryland has expanded in response to nearly five decades of urbanization (i.e., watershed IC increased from two to 27%).

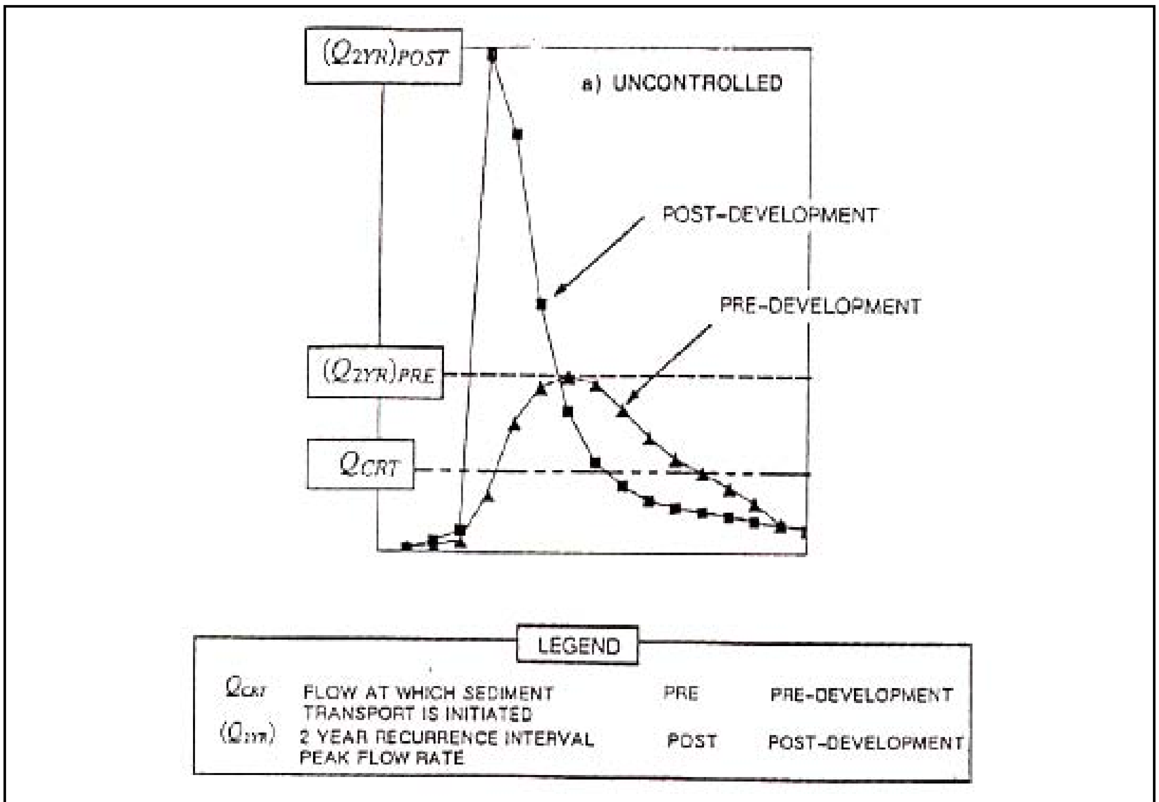


Figure 20: Increased Shear Stress from a Hydrograph (MacRae and Rowney, 1992)

Table 13: Research Review of Channel Enlargement and Sediment Transport in Urban Streams		
Reference	**Key Finding**	**Location**
% IC used as Indicator		
Caraco, 2000b	Reported enlargement in ratios of 1.5 to 2.2 for 10 stream reaches in Watts Branch and computed ultimate enlargement ratios of 2.0	MD
MacCrae and De Andrea, 1999	Introduced the concept of ultimate channel enlargement based on watershed IC and channel characteristics.	Ontario, TX
Morse, 2001	Demonstrated increased erosion rates with increases in IC (channels were generally of the same geomorphic type).	ME
Urbanization Used as Indicator		
Allen and Narramore, 1985	Enlargement ratios in two urban streams ranged from 1.7 to 2.4.	TX
Bledsoe, 2001	Reported that channel response to urbanization depends on other factors in addition to watershed IC including geology, vegetation, sediment and flow regimes.	N/A
Booth and Henshaw, 2001	Evaluated channel cross section erosion rates and determined that these rates vary based on additional factors including the underlying geology, age of development and gradient.	WA
Hammer, 1972	Enlargement ratios ranged from 0.7 to 3.8 in urban watersheds.	PA
Neller, 1989	Enlargement ratios in small urban catchments ranged from two to 7.19, the higher enlargement ratios were primarily from incision occurring in small channels.	Australia
Pizzuto *et al.*, 2000	Evaluated channel characteristics of paired urban and rural streams and demonstrated median bankfull cross sectional increase of 180%. Median values for channel sinuosity were 8% lower in urban streams; Mannings N values were found to be 10% lower in urban streams.	PA
Hession *et al.*, *in press*	Bankfull widths for urban streams were significantly wider than non-urban streams in 26 paired streams. Forested reaches were consistently wider than non-forested reaches in urban streams.	MD, DE, PA
Dartiguenave *et al.*, 1997	Bank erosion accounted for up to 75% of the sediment transport in urban watersheds.	TX
Trimble, 1997	Demonstrated channel enlargement over time in an urbanizing San Diego Creek; Bank erosion accounted for over 66% of the sediment transport.	CA

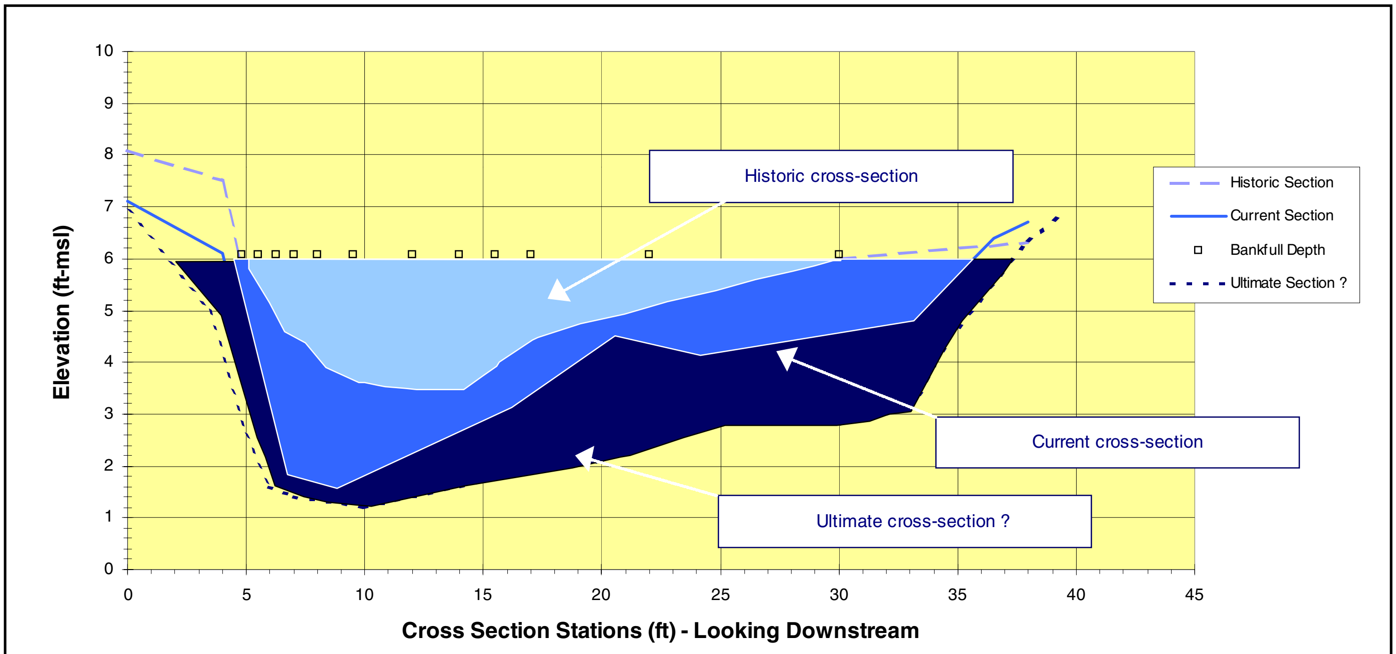


Figure 21: Stream Channel Enlargement in Watts Branch, MD 1950-2000 (Caraco, 2000b)

Some geomorphologists suggest that urban stream channels will reach an "ultimate enlargement" relative to pre-developed channels (MacRae and DeAndrea, 1999) and that this can be predicted based on watershed IC, age of development, and the resistance of the channel bed and banks. A relationship between ultimate stream channel enlargement and watershed IC has been developed for alluvial streams in Texas, Vermont and Maryland (Figure 22). Other geomorphologists such as Bledsoe (2001) and Booth and Henshaw (2001) contend that channel response to urbanization is more complex, and also depends on geology, grade control, stream gradient and other factors.

Channel incision is often limited by grade control caused by bedrock, cobbles, armored substrates, bridges, culverts and pipelines. These features can impede the downward erosion of the stream channel and thereby limit the incision process. Stream incision can become severe in streams that have softer substrates such as sand, gravel and clay (Booth, 1990). For example, Allen and Narramore (1985) showed that channel enlargement in chalk channels was 12 to 67% greater than in shale channels near Dallas, Texas. They attributed the differences to the softer substrate, greater velocities and higher shear stress in the chalk channels.

Neller (1989) and Booth and Henshaw (2001) also report that incised urban stream channels possess cross-sectional areas that are larger than would be predicted based on watershed area or discharge alone. This is due to the fact that larger floods are often contained within the stream channel rather than the floodplain. Thus, incised channels often result in greater erosion and geomorphic change. In general, stream conditions that can foster incision include erodible substrates, moderate to high stream gradients, and an absence of grade control features.

Channel widening occurs more frequently when streams have grade control and the stream has cut into its bank, thereby expanding its cross-sectional area. Urban stream channels often have artificial grade controls caused by frequent culverts and road crossings. These grade controls often cause localized sediment deposition that can reduce the capacity of culverts and bridge crossings to pass flood waters.

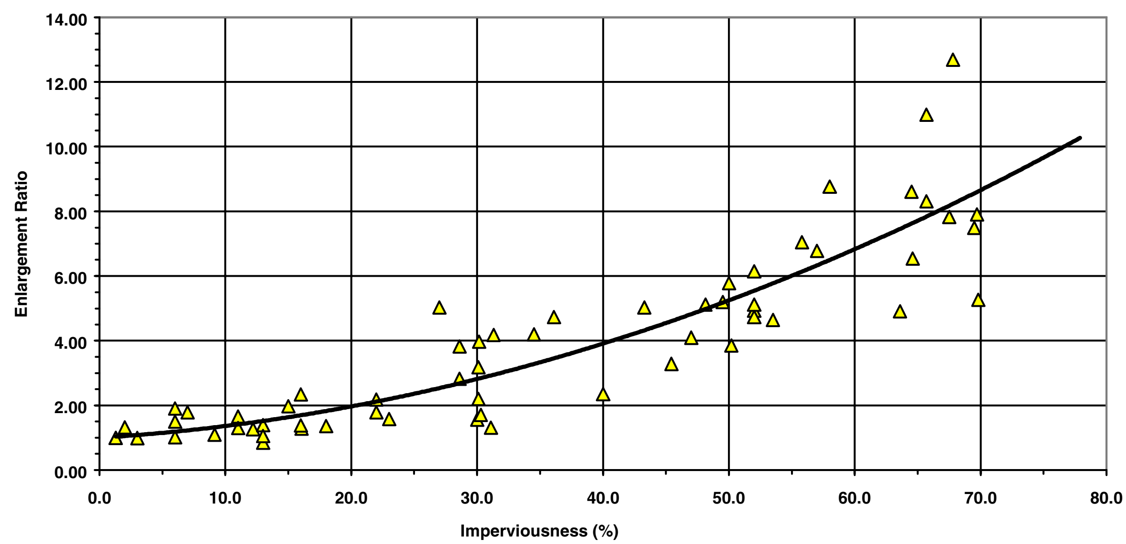


Figure 22: Ultimate Channel Enlargement in MD, UT and TX Alluvial Streams (MacRae and DeAndrea, 1999 and CWP, 2001b)

The loss of flood plain and riparian vegetation has been strongly associated with watershed urbanization (May *et al.*, 1997). A few studies have shown that the loss of riparian trees can result in increased erosion and channel migration rates (Beeson and Doyle, 1995 and Allmendinger *et al.*, 1999). For example, Beeson and Doyle (1995) found that meander bends with vegetation were five times less likely to experience significant erosion from a major flood than non-vegetated meander bends.Hession *et al.* (in press) observed that forested reaches consistently had greater bankfull widths than non-forested reaches in a series of urban streams in Pennsylvania, Maryland and Delaware.

3.2.2 Effect of Channel Enlargement on Sediment Yield

Regardless of whether a stream incises, widens, or does both, it will greatly increase sediment transport from the watershed due to erosion. Urban stream research conducted in California and Texas suggests that 60 to 75% of the sediment yield of urban watersheds can be derived from channel erosion (Trimble, 1997 and Dartingunave *et al.*, 1997) This can be compared to estimates for rural streams where channel erosion accounts for only five to 20% of the annual sediment yield (Collins *et al.*, 1997 and Walling and Woodward, 1995).

Some geomorphologists speculate that urban stream channels will ultimately adjust to their post-development flow regime and sediment supply. Finkenbine *et al.* (2000) observed these conditions in Vancouver streams, where study streams eventually stabilized two decades after the watersheds were fully developed. In older urban streams, reduced sediment transport can be expected when urbanization has been completed. At this point, headwater stream channels are replaced by storm drains and pipes, which can transport less sediment. The lack of available sediment may cause downstream channel erosion, due to the diminished sediment supply found in the stream.

3.3 Effect on Composite Measures of Stream Habitat

Composite measures of stream habitat refer to assessments such as EPA's Habitat Rapid Bioassessment Protocol (RBP) that combine multiple habitat elements into a single score or index (Barbour *et al.*, 1999). For example, the RBP requires visual assessment of 10 stream habitat elements, including embeddedness, epifaunal substrate quality, velocity/depth regime, sediment deposition, channel flow status, riffle frequency, bank stabilization, streambank vegetation and riparian vegetation width. Each habitat element is qualitatively scored on a 20 point scale, and each element is weighted equally to derive a composite score for the stream reach.

To date, several studies have found a relationship between declining composite habitat indicator scores and increasing watershed IC in different eco-regions of the United States. A typical pattern in the composite habitat scores is provided for headwater streams in Maine (Morse, 2001; Figure 23). This general finding has been reported in the mid-Atlantic, Northeast and the Northwest (Black and Veatch, 1994; Booth and Jackson, 1997; Hicks and Larson, 1997; Maxted and Shaver, 1997; Morse, 2001; Stranko and Rodney, 2001).

However, other researchers have found a much weaker relationship between composite habitat scores and watershed IC. Wang and his colleagues (2001) found that composite habitat scores were not correlated with watershed IC in Wisconsin streams, although it was correlated with individual habitat elements, such as streambank erosion. They noted that many agricultural and rural streams had fair to poor composite habitat scores, due to poor riparian management and sediment deposition. The same basic conclusion was also reported for streams of the Maryland Piedmont (MNCPPC, 2000).

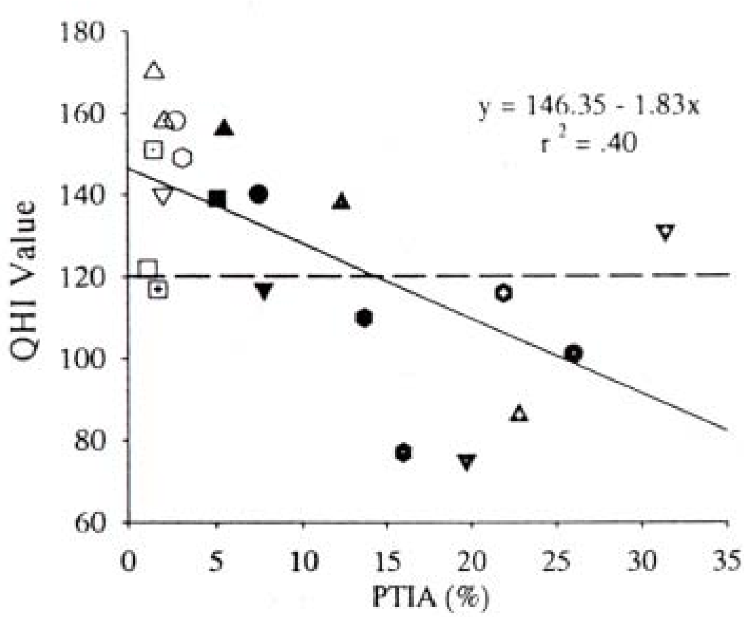


Figure 23: Relationship Between Habitat Quality and IC in Maine Streams (Morse, 2001)

3.4 Effect on Individual Elements of Stream Habitat

Roughly a dozen studies have examined the effect of watershed development on the degradation of individual stream habitat features such as bank stability, embeddedness, riffle/pool quality, and loss of LWD (Table 14). Much of this data has been acquired from the Pacific Northwest, where the importance of such habitat for migrating salmon has been a persistent management concern.

3.4.1 Bank Erosion and Bank Stability

It is somewhat surprising that we could only find one study that related bank stability or bank erosion to watershed IC. Conducted by Booth (1991) in the streams of the Puget Sound lowlands, the study reported that stream banks were consistently rated as stable in watersheds with less than 10% IC, but became progressively more unstable above this threshold. Dozens of stream assessments have found high rates of bank erosion in urban streams, but none, to our knowledge, has systematically related the prevalence or severity of bank erosion to watershed IC. As noted earlier, this may reflect the lack of a universally recognized method to measure comparative bank erosion in the field.

3.4.2 Embeddedness

Embeddedness is a term that describes the extent to which the rock surfaces found on the stream bottom are filled in with sand, silts and clay. In a healthy stream, the interstitial pores between cobbles, rock and gravel generally lack fine sediments, and are an active habitat zone and detrital processing area. The increased sediment transport in urban streams can rapidly fill up these pores in a process known as embedding. Normally, embeddedness is visually measured in riffle zones of streams. Riffles tend to be an important habitat for aquatic insects and fish (such as darters and sculpins). Clean stream substrates are also critical to trout and salmon egg incubation and embryo development. May *et al.* (1997) demonstrated that the percent of fine sediment particles in riffles generally increased with watershed IC (Figure 24). However, Finkenbine *et al.* (2000) reported that embeddedness eventually decreased slightly after watershed land use and sediment transport had stabilized for 20 years.

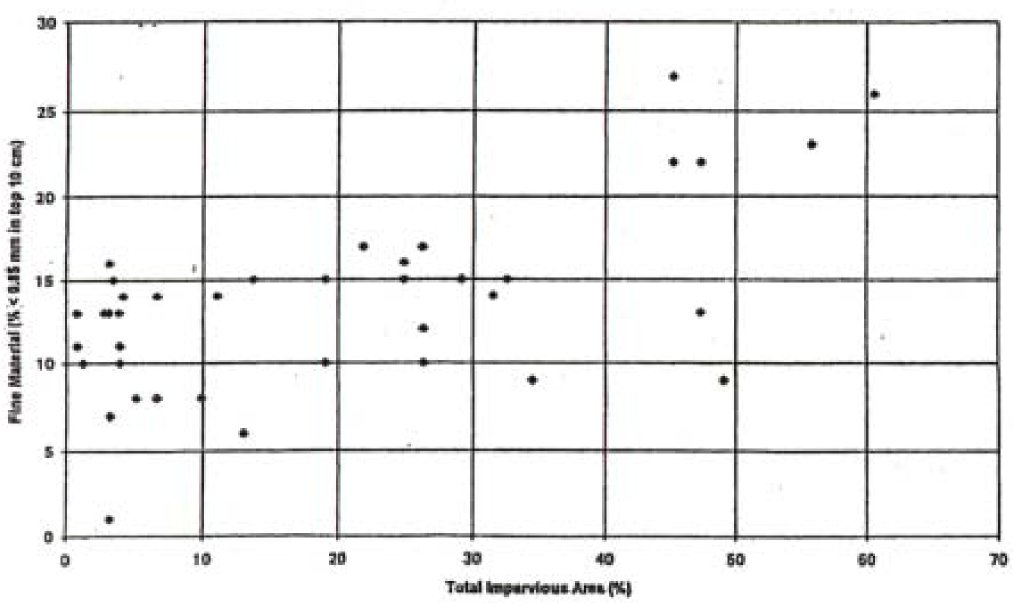


Figure 24: Fine Material Sediment Deposition as a Function of IC in Pacific Northwest Streams (Horner *et al.*, 1997)

Table 14: Research Review of Changes in Urban Stream Habitat		
Reference	**Key Finding**	**Location**
% IC Used as Indicator		
Black & Veatch, 1994	Habitat scores were ranked as poor in five subwatersheds that had greater than 30% IC.	MD
Booth and Jackson, 1997	Increase in degraded habitat conditions with increases in watershed IC.	WA
Hicks and Larson, 1997	Reported a reduction in composite stream habitat indices with increasing watershed IC.	MA
May *et al.*, 1997	Composite stream habitat declined most rapidly during the initial phase of the watershed urbanization, when percent IC exceeded the 5-10% range.	WA
Stranko and Rodney, 2001	Composite index of stream habitat declined with increasing watershed IC in coastal plain streams.	MD
Wang *et al.*, 2001	Composite stream habitat scores were not correlated with watershed IC in 47 small watersheds, although channel erosion was. Non-urban watersheds were highly agricultural and often lacked riparian forest buffers.	WI
MNCPPC, 2000	Reported that stream habitat scores were not correlated with IC in suburban watersheds.	MD
Morse, 2001	Composite habitat values tended to decline with increases in watershed IC.	ME
Booth, 1991	Channel stability and fish habitat quality declined rapidly after 10% watershed IC.	WA
Booth *et al.*, 1997	Decreased LWD with increased IC.	PNW
Finkenbine *et al.*, 2000	LWD was scarce in streams with greater than 20% IC in Vancouver.	B.C.
Horner & May, 1999	When IC levels were >5%, average LWD densities fell below 300 pieces/kilometer.	PNW
Horner *et al.*, 1997	Interstitial spaces in streambed sediments begin to fill with increasing watershed IC.	PNW
Urbanization Used as Indicator		
Dunne and Leopold, 1978	Natural channels replaced by storm drains and pipes; increased erosion rates observed downstream.	MD
May *et al.*, 1997	Forested riparian corridor width declines with increased watershed IC.	PNW
MWCOG, 1992	Fish blockages caused by bridges and culverts noted in urban watersheds.	D.C.
Pizzuto *et al.*, 2000	Urban streams had reduced pool depth, roughness, and sinuosity, compared to rural streams; Pools were 31% shallower in urban streams compared to non-urban ones.	PA
Richey, 1982	Altered pool/riffle sequence observed in urban streams.	WA
Scott *et al.*, 1986	Loss of habitat diversity noted in urban watersheds.	PNW
Spence *et al.*, 1996	Large woody debris is important for habitat diversity and anadromous fish.	PNW

3.4.3 Large Woody Debris (LWD)

LWD is a habitat element that describes the approximate volume of large woody material (< four inches in diameter) found in contact with the stream. The presence and stability of LWD is an important habitat parameter in streams. LWD can form dams and pools, trap sediment and detritus, stabilize stream channels, dissipate flow energy, and promote habitat complexity (Booth *et al.,* 1997). LWD creates a variety of pool features (plunge, lateral, scour and backwater); short riffles; undercut banks; side channels; and a range of water depths (Spence *et al.*, 1996). Urban streams tend to have a low supply of LWD, as increased stormwater flows transport LWD and clears riparian areas. Horner *et al.* (1997) presents evidence from Pacific Northwest streams that LWD decreases in response to increasing watershed IC (Figure 25).

3.4.4 Changes in Other Individual Stream Parameters

One of the notable changes in urban stream habitat is a decrease in pool depth and a general simplification of habitat features such as pools, riffles and runs. For example, Richey (1982) and Scott *et al.* (1986) reported an increase in the prevalence of glides and a corresponding altered riffle/pool sequence due to urbanization. Pizzuto *et al.* (2000) reported a median 31% decrease in pool depth in urban streams when compared to forested streams. Pizzuto *et al.* also reported a modest decrease in channel sinuosity and channel roughness in the same urban streams in Pennsylvania.

Several individual stream habitat parameters appear to have received no attention in urban stream research to date. These parameters include riparian shading, wetted perimeter, various measures of velocity/depth regimes, riffle frequency, and sediment deposition in pools. More systematic monitoring of these individual stream habitat parameters may be warranted.

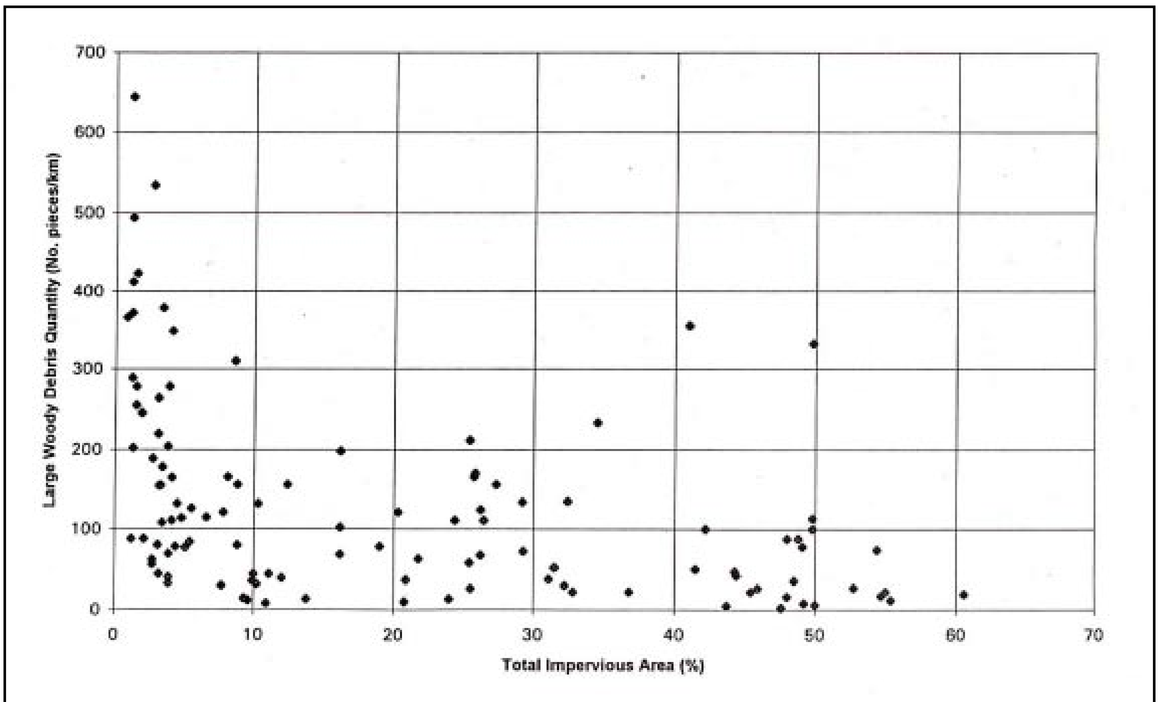


Figure 25: LWD as a Function of IC in Puget Sound Streams (Horner *et al.*, 1997)

3.5 Increased Stream Warming

IC directly influences our local weather in urban areas. This effect is obvious to anyone walking across a parking lot on a hot summer day, when temperatures often reach a scorching 110 to 120 degrees F. Parking lots and other hard surfaces tend to absorb solar energy and release it slowly. Furthermore, they lack the normal cooling properties of trees and vegetation, which act as natural air conditioners. Finally, urban areas release excess heat as a result of the combustion of fossil fuels for heating, cooling and transportation. As a result, highly urban areas tend to be much warmer than their rural counterparts and are known as urban heat islands. Researchers have found that summer temperatures tend to be six to eight degrees F warmer in the summer and two to four degrees F warmer during the winter months.

Water temperature in headwater streams is strongly influenced by local air temperatures. Summer temperatures in urban streams have been shown to increase by as much as five to 12 degrees F in response to watershed development (Table 15). Increased water temperatures can preclude temperature-sensitive species from being able to survive in urban streams. Figure 26 shows the stream warming phenomenon in small headwater streams in the Maryland Piedmont.

Galli (1990) reported that stream temperatures throughout the summer increased in urban watersheds. He monitored five headwater streams in the Maryland Piedmont with different levels of IC. Each urban stream had mean temperatures that were consistently warmer than a forested reference stream, and stream warming appeared to be a direct function of watershed IC. Other factors, such as lack of riparian cover and the presence of ponds, were also demonstrated to amplify stream warming, but the primary contributing factor appeared to be watershed IC.

Johnson (1995) studied how stormwater influenced an urban trout stream in Minnesota and reported up to a 10 degree F increase in stream water temperatures after summer storm events. Paul *et al.* (2001) evaluated stream temperatures for 30 subwatersheds to the Etowah River in Georgia, which ranged from five to 61% urban land. They found a correlation between summer daily mean water temperatures and the percentage of urban land in a subwatershed.

Table 15: Research Review of Thermal Impacts in Urban Streams

Reference	Key Finding	Location
%IC Used as Indicator		
Galli, 1990	Increase in stream temperatures of five to 12 degrees Fahrenheit in urban watersheds; stream warming linked to IC.	MD
Urbanization Used as Indicator		
Johnson, 1995	Up to 10 degrees Fahrenheit increases in stream temperatures after summer storm events in an urban area	MN
LeBlanc *et al.*, 1997	Calibrated a model predicting stream temperature increase as a result of urbanization	Ontario
MCDEP, 2000	Monitoring effect of urbanization and stormwater ponds on stream temperatures revealed stream warming associated with urbanization and stormwater ponds	MD
Paul *et al.*, 2001	Daily mean stream temperatures in summer increased with urban land use	GA

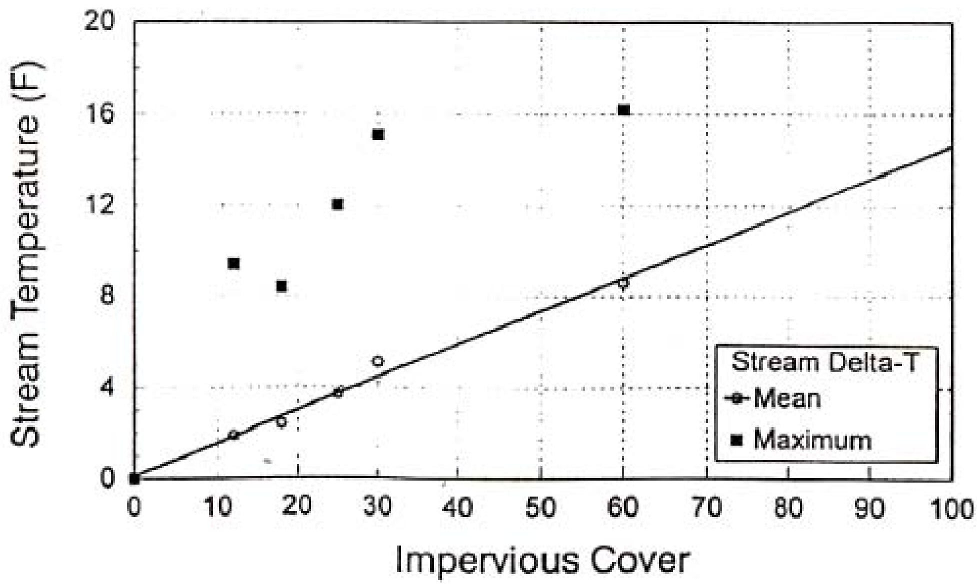


Figure 26: Stream Temperature Increase in Response to IC in Maryland Piedmont Streams (Galli, 1990)

Discharges from stormwater ponds can also contribute to stream warming in urban watersheds. Three studies highlight the temperature increase that can result from stormwater ponds. A study in Ontario found that baseflow temperatures below wet stormwater ponds increased by nine to 18 degrees F in the summer (SWAMP, 2000a, b). Oberts (1997) also measured change in the baseflow temperature as it flowed through a wetland/wet pond system in Minnesota. He concluded that the temperature had increased by an average of nine degrees F during the summer months. Galli (1988) also observed a mean increase of two to 10 degrees F in four stormwater ponds located in Maryland.

3.6 Alteration of Stream Channel Networks

Urban stream channels are often severely altered by man. Channels are lined with rip rap or concrete, natural channels are straightened, and first order and ephemeral streams are enclosed in storm drain pipes. From an engineering standpoint, these modifications rapidly convey flood waters downstream and locally stabilize stream banks. Cumulatively, however, these modifications can have a dramatic effect on the length and habitat quality of headwater stream networks.

3.6.1 Channel Modification

Over time, watershed development can alter or eliminate a significant percentage of the perennial stream network. In general, the loss of stream network becomes quite extensive when watershed IC exceeds 50%. This loss is striking when pre- and post-development stream networks are compared (Figure 27). The first panel illustrates the loss of stream network over time in a highly urban Northern Virginia watershed; the second panel shows how the drainage network of Rock Creek has changed in response to watershed development.

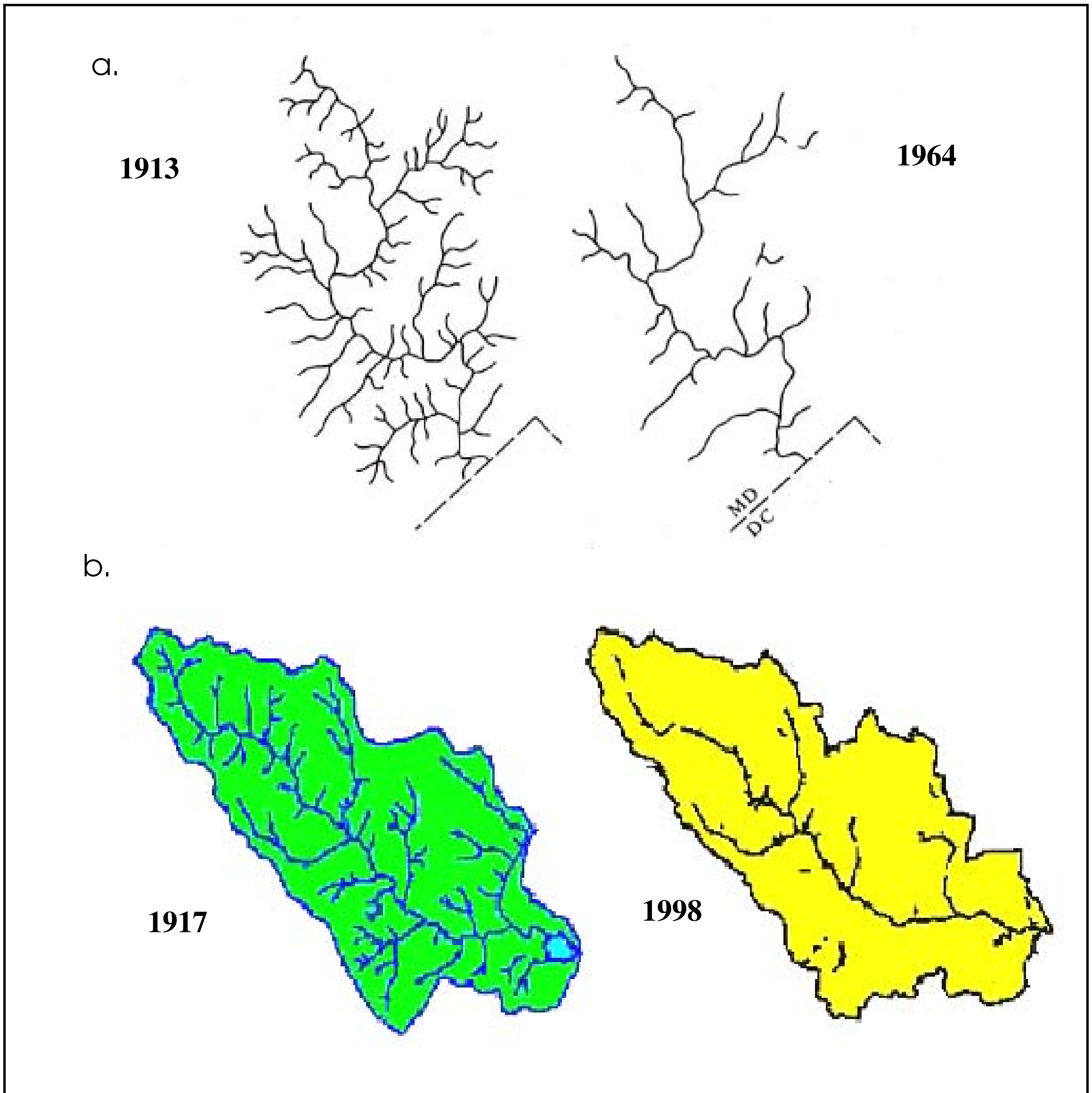


Figure 27: a. Drainage Network of Rock Creek, D.C. (Dunne and Leopold, 1978) and b. Drainage Network of Four Mile Run, VA Before and After Urbanization (NVRC, 2001)

In a national study of 269 gaged urban watersheds, Sauer *et al.* (1983) observed that channelization and channel hardening were important watershed variables that control peak discharge rates. The channel modifications increase the efficiency with which runoff is transported through the stream channel, increasing critical shear stress velocities and causing downstream channel erosion.

3.6.2 Barriers to Fish Migration

Infrastructure such as bridges, dams, pipelines and culverts can create partial or total barriers to fish migration and impair the ability of fish to move freely in a watershed. Blockages can have localized effects on small streams where non-migratory fish species can be prevented from re-colonizing upstream areas after acutely toxic events. The upstream movement of anadromous fish species such as shad, herring, salmon and steelhead can also be blocked by these barriers. Figure 28 depicts the prevalence of fish barriers in the Anacostia Watershed (MWCOG, 1992).

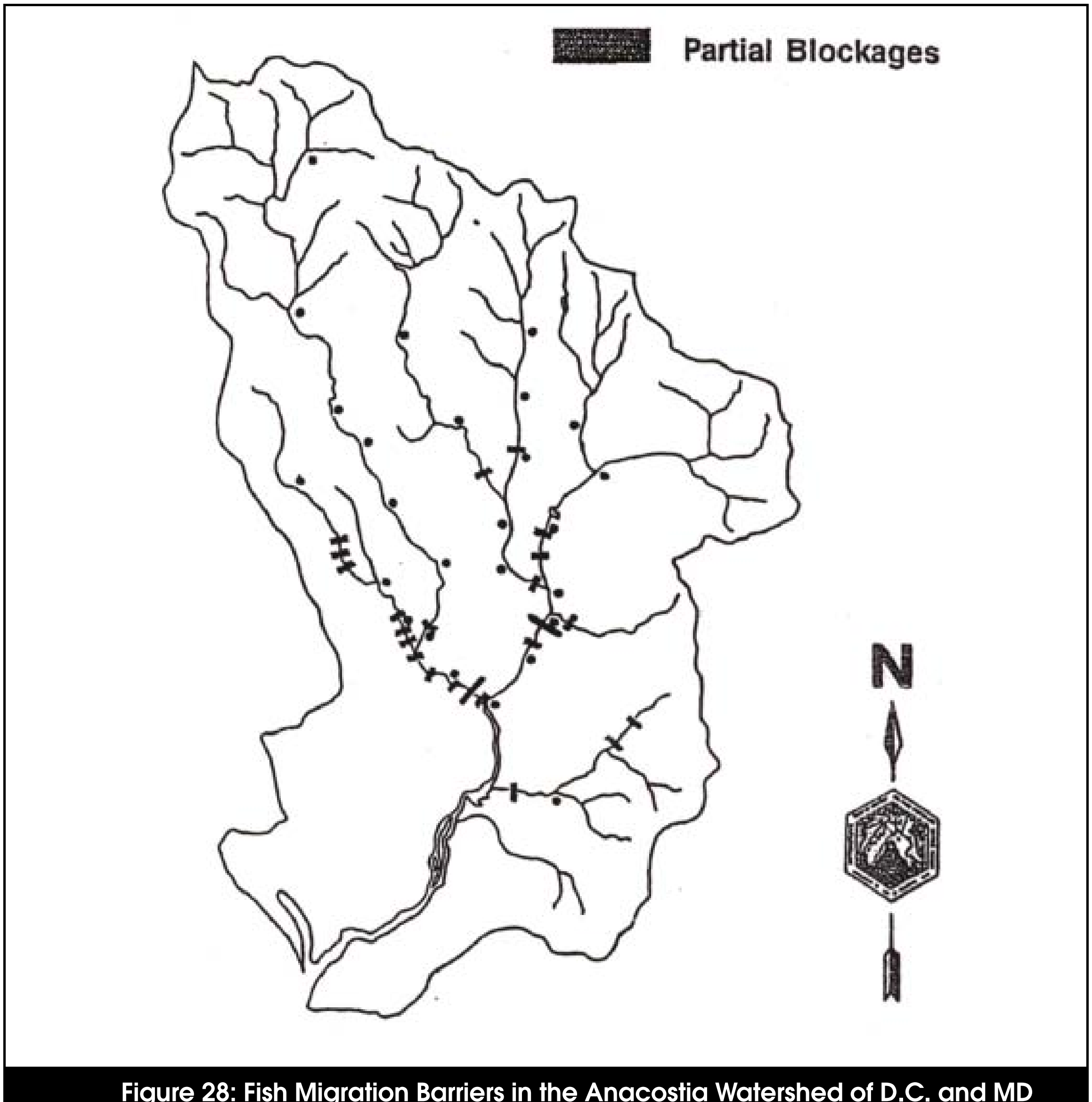


Figure 28: Fish Migration Barriers in the Anacostia Watershed of D.C. and MD (MWCOG, 1992)

3.7 Conclusion

Watershed development and the associated increase in IC have been found to significantly degrade the physical habitat of urban streams. In alluvial streams, the effects of channel enlargement and sediment transport can be severe at relatively low levels of IC (10 to 20%). However, the exact response of any stream is also contingent upon a combination of other physical factors such as geology, vegetation, gradient, the age of development, sediment supply, the use and design of stormwater treatment practices, and the extent of riparian buffers (Bledsoe, 2001).

Despite the uncertainty introduced by these factors, the limited geomorphic research to date suggests that physical habitat quality is almost always degraded by higher levels of watershed IC. Even in bedrock-controlled channels, where sediment transport and channel enlargement may not be as dramatic, researchers have noted changes in stream habitat features, such as embeddedness, loss of LWD, and stream warming.

Overall, the following conclusions can be made about the influence of watershed development on the physical habitat of urban streams:

- The major changes in physical habitat in urban streams are caused by the increased frequency and duration of bankfull and sub-bankfull discharges, and the attendant changes in sediment supply and transport. As a consequence, many urban streams experience significant channel enlargement. Generally, channel enlargement is most evident in alluvial streams.

- Typical habitat changes observed in urban streams include increased embeddedness, reduced supply of LWD, and simplification of stream habitat features such as pools, riffles and runs, as well as reduced channel sinuosity.

- Stream warming is often directly linked to watershed development, although more systematic subwatershed sampling is needed to precisely predict the extent of warming.

- Channel straightening, hardening and enclosure and the creation of fish barriers are all associated with watershed development. More systematic research is needed to establish whether these variables can be predicted based on watershed IC.

- In general, stream habitat diminishes at about 10% watershed IC, and becomes severely degraded beyond 25% watershed IC.

While our understanding of the relationship between stream habitat features and watershed development has improved in recent years, the topic deserves greater research in three areas. First, more systematic monitoring of composite habitat variables needs to be conducted across the full range of watershed IC. In particular, research is needed to define the approximate degree of watershed IC where urban streams are transformed into urban drainage systems.

Second, additional research is needed to explore the relationship between watershed IC and individual and measurable stream habitat parameters, such as bank erosion, channel sinuosity, pool depth and wetted perimeter. Lastly, more research is needed to determine if watershed treatment such as stormwater practices and stream buffers can mitigate the impacts of watershed IC on stream habitat. Together, these three research efforts could provide a technical foundation to develop a more predictive model of how watershed development influences stream habitat.

Chapter 4: Water Quality Impacts of Impervious Cover

This chapter presents information on pollutant concentrations found in urban stormwater runoff based on a national and regional data assessment for nine categories of pollutants. Included is a description of the Simple Method, which can be used to estimate pollutant loads based on the amount of IC found in a catchment or subwatershed. This chapter also addresses specific water quality impacts of stormwater pollutants and explores research on the sources and source areas of stormwater pollutants.

This chapter is organized as follows:

4.1 Introduction

Streams are usually the first aquatic system to receive stormwater runoff, and their water quality can be compromised by the pollutants it contains. Stormwater runoff typically contains dozens of pollutants that are detectable at some concentration, however small. Simply put, any pollutant deposited or derived from an activity on land will likely end up in stormwater runoff, although certain pollutants are consistently more likely to cause water quality problems in receiving waters. Pollutants that are frequently found in stormwater runoff can be grouped into nine broad categories: sediment, nutrients, metals, hydrocarbons, bacteria and pathogens, organic carbon, MTBE, pesticides, and deicers.

The impact that stormwater pollutants exert on water quality depends on many factors, including concentration, annual pollutant load, and category of pollutant. Based on nationally reported concentration data, there is considerable variation in stormwater pollutant concentrations. This variation has been at least partially attributed to regional differences, including rainfall and snowmelt. The volume and regularity of rainfall, the length of snow accumulation, and the rate of snowmelt can all influence stormwater pollutant concentrations.

The annual pollutant load can have long-term effects on stream water quality, and is particularly important information for stormwater managers to have when dealing with non-point source pollution control. The Simple Method is a model developed to estimate the pollutant load for chemical pollutants, assuming that the annual pollutant load is a function of IC. It is an effective method for determining annual sediment, nutrient, and trace metal loads. It cannot always be applied to other stormwater pollutants, since they are not always correlated with IC.

The direct water quality impact of stormwater pollutants also depends on the type of pollutant, as different pollutants impact streams differently. For example, sediments affect stream habitat and aquatic biodiversity; nutrients cause eutrophication; metals, hydrocarbons, deicers, and MTBE can be toxic to aquatic life; and organic carbon can lower dissolved oxygen levels.

The impact stormwater pollutants have on water quality can also directly influence human uses and activities. Perhaps the pollutants of greatest concern are those with associated public health impacts, such as bacteria and pathogens. These pollutants can affect the availability of clean drinking water and limit consumptive recreational activities, such as swimming or fishing. In extreme situations, these pollutants can even limit contact recreational activities such as boating and wading.

It should be noted that although there is much research available on the effects of urbanization on water quality, the majority has not been focused on the impact on streams, but on the response of lakes, reservoirs, rivers and estuaries. It is also important to note that not all pollutants are equally represented in monitoring conducted to date. While we possess excellent monitoring data for sediment, nutrients and trace metals, we have relatively little monitoring data for pesticides, hydrocarbons, organic carbon, deicers, and MTBE.

4.2 Summary of National and Regional Stormwater Pollutant Concentration Data

4.2.1 National Data

National mean concentrations of typical stormwater pollutants are presented in Table 16. National stormwater data are compiled from the Nationwide Urban Runoff Program (NURP), with additional data obtained from the U.S.Geological Survey (USGS), as well as initial stormwater monitoring conducted for EPA's National Pollutant Discharge Elimination System (NPDES) Phase I stormwater program.

In most cases, stormwater pollutant data is reported as an event mean concentration (EMC), which represents the average concentration of the pollutant during an entire stormwater runoff event.

When evaluating stormwater EMC data, it is important to keep in mind that regional EMCs can differ sharply from the reported national pollutant EMCs. Differences in EMCs between regions are often attributed to the variation in the amount and frequency of rainfall and snowmelt.

4.2.2 Regional Differences Due to Rainfall

The frequency of rainfall is important, since it influences the accumulation of pollutants on IC that are subsequently available for wash-off during storm events. The USGS developed a national stormwater database encompassing 1,123 storms in 20 metropolitan areas and used it as the primary data source to define regional differences in stormwater EMCs. Driver (1988) performed regression analysis to determine which factors had the greatest influence on stormwater EMCs and determined that annual rainfall depth was the best overall predictor. Driver grouped together stormwater EMCs based on the depth of average annual rainfall, and Table 17 depicts the regional rainfall groupings and general trends for each region. Table 18 illustrates the distribution of stormwater EMCs for a range of rainfall regions from 13 local studies, based on other monitoring studies. In general, stormwater EMCs for nutrients, suspended sediment and metals tend to be higher in arid and semi-arid

Table 16: National EMCs for Stormwater Pollutants

Pollutant	Source	EMCs		Number of Events
		Mean	Median	
Sediments (mg/l)				
TSS	(1)	78.4	54.5	3047
Nutrients (mg/l)				
Total P	(1)	0.32	0.26	3094
Soluble P	(1)	0.13	0.10	1091
Total N	(1)	2.39	2.00	2016
TKN	(1)	1.73	1.47	2693
Nitrite & Nitrate	(1)	0.66	0.53	2016
Metals (Fg/l)				
Copper	(1)	13.4	11.1	1657
Lead	(1)	67.5	50.7	2713
Zinc	(1)	162	129	2234
Cadmium	(1)	0.7	N/R	150
Chromium	(4)	4	7	164
Hydrocarbons (mg/l)				
PAH	(5)	3.5	N/R	N/R
Oil and Grease	(6)	3	N/R	N/R
Bacteria and Pathogens (colonies/ 100ml)				
Fecal Coliform	(7)	15,038	N/R	34
Fecal Streptococci	(7)	35,351	N/R	17
Organic Carbon (mg/l)				
TOC	(11)	17	15.2	19 studies
BOD	(1)	14.1	11.5	1035
COD	(1)	52.8	44.7	2639
MTBE	(8)	N/R	1.6	592
Pesticides (Fg/l)				
Diazinon	(10)	N/R	0.025	326
	(2)	N/R	0.55	76
Chlorpyrifos	(10)	N/R	N/R	327
Atrazine	(10)	N/R	0.023	327
Prometon	(10)	N/R	0.031	327
Simazine	(10)	N/R	0.039	327
Chloride (mg/l)				
Chloride	(9)	N/R	397	282

Sources: [1] Smullen and Cave, 1998; [2] Brush et al., 1995; [3] Baird et al., 1996; [4] Bannerman et al., 1996; [5] Rabanal and Grizzard, 1995; [6] Crunkilton et al., 1996; [7] Schueler, 1999; [8] Delzer, 1996; [9] Environment Canada, 2001; [10] USEPA, 1998; [11] CWP, 2001a   N/R - Not Reported

Table 17: Regional Groupings by Annual Rainfall Amount (Driver, 1988)

Region	Annual Rainfall	States Monitored	Concentration Data
Region I: Low Rainfall	<20 inches	AK, CA, CO, NM, UT	Highest mean and median values for Total N, Total P, TSS and COD
Region II: Moderate Rainfall	20 40 inches	HA, IL, MI, MN, MI, NY, TX, OR, OH, WA, WI	Higher mean and median values than Region III for TSS, dissolved phosphorus and cadmium
Region III: High Rainfall	>40 inches	FL, MD, MA, NC, NH, NY, TX, TN, AR	Lower values for many parameters likely due to the frequency of storms and the lack of build up in pollutants

regions and tend to decrease slightly when annual rainfall increases (Table 19).

It is also hypothesized that a greater amount of sediment is eroded from pervious surfaces in arid or semi-arid regions than in humid regions due to the sparsity of protective vegetative cover. Table 19 shows that the highest concentrations of total suspended solids were recorded in regions with least rainfall. In addition, the chronic toxicity standards for several metals are most frequently exceeded during low rainfall regions (Table 20).

4.2.3 Cold Region Snowmelt Data

In colder regions, snowmelt can have a significant impact on pollutant concentrations. Snow accumulation in winter coincides with pollutant build-up; therefore, greater concentrations of pollutants are measured during snowmelt events. Sources of snowpack pollution in urban areas include wet and dry atmospheric deposition, traffic emissions, urban litter, deteriorated infrastructure, and deicing chemicals and abrasives (WERF, 1999).

Oberts *et al.* (1989) measured snowmelt pollutants in Minnesota streams and found that as much as 50% of annual sediment, nutrient, hydrocarbon and metal loads could be attributed to snowmelt runoff during late winter and early spring. This trend probably applies to any region where snow cover persists through much of the winter. Pollutants accumulate in the snowpack and then contribute high concentrations during snowmelt runoff. Oberts (1994) described four types of snowmelt runoff events and the resulting pollutant characteristics (Table 21).

A typical hydrograph for winter and early spring snow melts in a northern cold climate is portrayed in Figure 29. The importance of snowpack melt on peak runoff during March 1989 can clearly be seen for an urban watershed located in St. Paul, Minnesota.

Major source areas for snowmelt pollutants include snow dumps and roadside snowpacks. Pollutant concentrations in snow dumps can be as much as five times greater than typical stormwater pollutant concentrations (Environment Canada, 2001). Snow dumps and packs accumulate pollutants over the winter months and can release them during a few rain or snow melt events in the early spring. High levels of chloride, lead, phosphorus, biochemical oxygen demand, and total suspended solids have been reported in snow pack runoff (La Barre *et al*, 1973; Oliver *et al.*, 1974; Pierstorff and Bishop, 1980; Scott and Wylie, 1980; Van Loon, 1972).

Atmospheric deposition can add pollutants to snow piles and snowpacks. Deposited pollutants include trace metals, nutrients and particles that are primarily generated by fossil fuel combustion and industrial emissions (Boom and Marsalek, 1988; Horkeby and Malmqvist, 1977; Malmqvist, 1978; Novotny and Chester, 1981; Schrimpff and Herrman, 1979).

Table 18: Stormwater Pollutant Event Mean Concentration for Different U.S. Regions (Units: mg/l, except for metals which are in *F*g/l)

		Region I - Low Rainfall				Region II - Moderate Rainfall			Region III - High Rainfall				Snow
	National	Phoenix, AZ	San Diego, CA	Boise, ID	Denver, CO	Dallas, TX	Marquette, MI	Austin, TX	MD	Louisville, KY	GA	FL	MN
Reference	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(11)	(12)
Annual Rainfall (in.)	N/A	7.1"	10"	11"	15"	28"	32"	32"	41"	43"	51"	52"	N/R
Number of Events	3000	40	36	15	35	32	12	N/R	107	21	81	N/R	49
Pollutant													
TSS	78.4	227	330	116	242	663	159	190	67	98	258	43	112
Total N	2.39	3.26	4.55	4.13	4.06	2.70	1.87	2.35	N/R	2.37	2.52	1.74	4.30
Total P	0.32	0.41	0.7	0.75	0.65	0.78	0.29	0.32	0.33	0.32	0.33	0.38	0.70
Soluble P	0.13	0.17	0.4	0.47	N/R	N/R	0.04	0.24	N/R	0.21	0.14	0.23	0.18
Copper	14	47	25	34	60	40	22	16	18	15	32	1.4	N/R
Lead	68	72	44	46	250	330	49	38	12.5	60	28	8.5	100
Zinc	162	204	180	342	350	540	111	190	143	190	148	55	N/R
BOD	14.1	109	21	89	N/R	112	15.4	14	14.4	88	14	11	N/R
COD	52.8	239	105	261	227	106	66	98	N/R	38	73	64	112

Sources: Adapted from Caraco, 2000a: [1] Smullen and Cave, 1998; [2] Lopes et al.; 1995; [3] Schiff, 1996; [4] Kjelstrom, 1995 (computed); [5] DRCOG, 1983, [6] Brush et al.*, 1995; [7] Steuer* et al.*, 1997; [8] Barrett* et al.*, 1995; [9] Barr, 1997; [10] Evaldi* et al.*, 1992; [11] Thomas and McClelland, 1995; [12] Oberts, 1994 N/R = Not Reported; N/A = Not Applicable*

Table 19: Mean and Median Nutrient and Sediment Stormwater Concentrations for Residential Land Use Based on Rainfall Regions (Driver, 1988)

Region	Total N (median)	Total P (median)	TSS (mean)
Region I: Low Rainfall	4	0.45	320
Region II: Moderate Rainfall	2.3	0.31	250
Region III: High Rainfall	2.15	0.31	120

Table 20: EPA 1986 Water Quality Standards and Percentage of Metal Concentrations Exceeding Water Quality Standards by Rainfall Region (Driver, 1988)

	Cadmium	Copper	Lead	Zinc
EPA Standards	10 *F*g/l	12 *F*g/l	32 *F*g/l	47 *F*g/l
	Percent Exceedance of EPA Standards			
Region I: Low Rainfall	1.5%	89%	97%	97%
Region II: Moderate Rainfall	0	78%	89%	85%
Region III: High Rainfall	0	75%	91%	84%

Table 21: Runoff and Pollutant Characteristics of Snowmelt Stages (Oberts, 1994)

Snowmelt Stage	Duration /Frequency	Runoff Volume	Pollutant Characteristics
Pavement	Short, but many times in winter	Low	Acidic, high concentrations of soluble pollutants; Chloride, nitrate, lead; total load is minimal
Roadside	Moderate	Moderate	Moderate concentrations of both soluble and particulate pollutants
Pervious Area	Gradual, often most at end of season	High	Dilute concentrations of soluble pollutants; moderate to high concentrations of particulate pollutants depending on flow
Rain-on-Snow	Short	Extreme	High concentrations of particulate pollutants; moderate to high concentrations of soluble pollutants; high total load

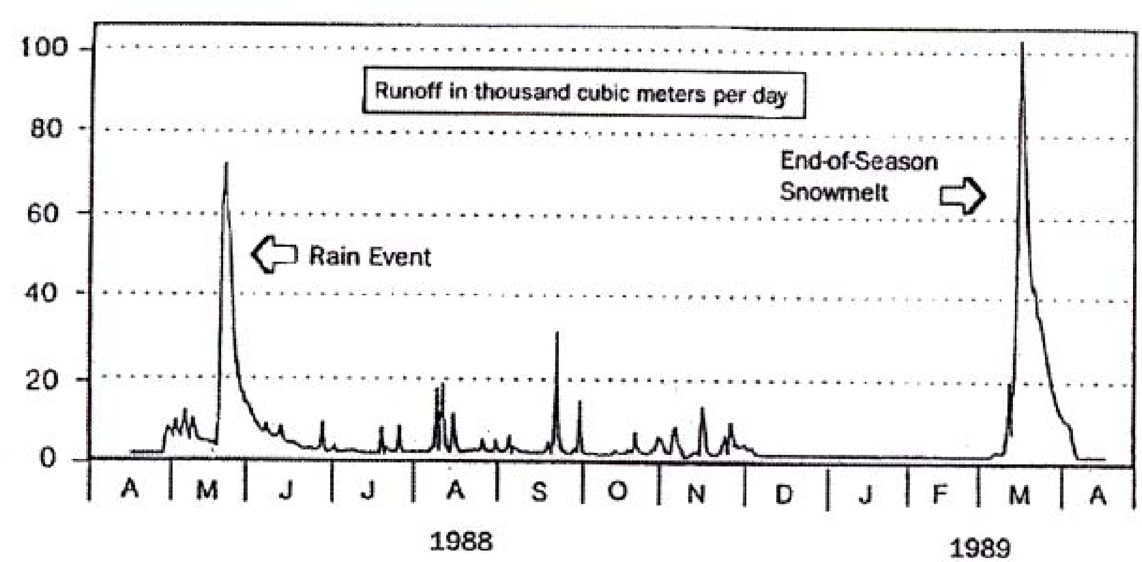


Figure 29: Snowmelt Runoff Hydrograph for Minneapolis Stream (Oberts, 1994)

4.3 Relationship Between Pollutant Loads and IC: The Simple Method

Urban stormwater runoff contains a wide range of pollutants that can degrade downstream water quality. The majority of stormwater monitoring research conducted to date supports several generalizations. First, the unit area pollutant load delivered to receiving waters by stormwater runoff increases in direct proportion to watershed IC. This is not altogether surprising, since pollutant load is the product of the average pollutant concentration and stormwater runoff volume. Given that runoff volume increases in direct proportion to IC, pollutant loads must automatically increase when IC increases, as long the average pollutant concentration stays the same (or increases). This relationship is a central assumption in most simple and complex pollutant loading models (Bicknell *et al.*, 1993; Donigian and Huber, 1991; Haith *et al.*, 1992; Novotny and Chester, 1981; NVPDC, 1987; Pitt and Voorhees, 1989).

Recognizing the relationship between IC and pollutant loads, Schueler (1987) developed the "Simple Method" to quickly and easily estimate stormwater pollutant loads for small urban watersheds (see Figure 30). Estimates of pollutant loads are important to watershed managers as they grapple with costly decisions on non-point source control. The Simple Method is empirical in nature and utilizes the extensive regional and national database (Driscoll, 1983; MWCOG, 1983; USEPA, 1983). Figure 30 provides the basic equations to estimate pollutant loads using the Simple

Figure 30: The Simple Method - Basic Equations

The Simple Method estimates pollutant loads as the product of annual runoff volume and pollutant EMC, as:

$$(1)\ L = 0.226 * R * C * A$$

Where: L = Annual load (lbs), and:
R = Annual runoff (inches)
C = Pollutant concentration in stormwater, EMC (mg/l)
A = Area (acres)
0.226 = Unit conversion factor

For bacteria, the equation is slightly different, to account for the differences in units. The modified equation for bacteria is:

$$(2)\ L = 1.03 * 10^{-3} * R * C * A$$

Where: L = Annual load (Billion Colonies), and:
R = Annual runoff (inches)
C = Bacteria concentration (#/100 ml)
A = Area (acres)
$1.03 * 10^{-3}$ = Unit conversion factor

Annual Runoff

The Simple Method calculates the depth of annual runoff as a product of annual runoff volume and a runoff coefficient (Rv). Runoff volume is calculated as:

$$(3)\ R = P * P_j * Rv$$

Where: R = Annual runoff (inches), and:
P = Annual rainfall (inches)
P_j = Fraction of annual rainfall events that produce runoff (usually 0.9)
Rv = Runoff coefficient

In the Simple Method, the runoff coefficient is calculated based on IC in the subwatershed. The following equation represents the best fit line for the data set (N=47, R^2=0.71).

$$(4)\ Rv=0.05+0.9Ia$$

Where: Rv = runoff coefficient, and:
Ia = Impervious fraction

Method. It assumes that loads of stormwater pollutants are a direct function of watershed IC, as IC is the key independent variable in the equation.

The technique requires a modest amount of information, including the subwatershed drainage area, IC, stormwater runoff pollutant EMCs, and annual precipitation. With the Simple Method, the investigator can either divide up land use into specific areas (i.e. residential, commercial, industrial, and roadway) and calculate annual pollutant loads for each land use, or utilize a generic urban land use. Stormwater pollutant EMC data can be derived from the many summary tables of local, regional, or national monitoring efforts provided in this chapter (e.g., Tables 16, 18, 22, 28, 30, 35, 36, 40, and 44). The model also requires different IC values for separate land uses within a subwatershed. Representative IC data from Cappiella and Brown (2001) were provided in Table 2 (Chapter 1).

Additionally, the Simple Method should not be used to estimate annual pollutant loads of deicers, hydrocarbons and MTBE, because they have not been found to be correlated with IC. These pollutants have been linked to other indicators. Chlorides, hydrocarbons and MTBE are often associated with road density and vehicle miles traveled (VMT). Pesticides are associated with turf area, and traffic patterns and "hotspots" have been noted as potential indicators for hydrocarbons and MTBE.

Limitations of the Simple Method

The Simple Method should provide reasonable estimates of changes in pollutant export resulting from urban development. However, several caveats should be kept in mind when applying this method.

The Simple Method is most appropriate for assessing and comparing the relative stormflow pollutant load changes from different land uses and stormwater treatment scenarios. The Simple Method provides estimates of storm pollutant export that are probably close to the "true" but unknown value for a development site, catchment, or subwatershed. However, it is very important not to overemphasize the precision of the load estimate obtained. For example, it would be inappropriate to use the Simple Method to evaluate relatively similar development scenarios (e.g., 34.3% versus 36.9% IC). The Simple Method provides a general planning estimate of likely storm pollutant export from areas at the scale of a development site, catchment or subwatershed. More sophisticated modeling is needed to analyze larger and more complex watersheds.

In addition, the Simple Method only estimates pollutant loads generated during storm events. It does not consider pollutants associated with baseflow during dry weather. Typically, baseflow is negligible or non-existent at the scale of a single development site and can be safely neglected. However, catchments and subwatersheds do generate significant baseflow volume. Pollutant loads in baseflow are generally low and can seldom be distinguished from natural background levels (NVPDC, 1979).

Consequently, baseflow pollutant loads normally constitute only a small fraction of the total pollutant load delivered from an urban area. Nevertheless, it is important to remember that the load estimates refer only to storm event derived loads and should not be confused with the total pollutant load from an area. This is particularly important when the development density of an area is low. For example, in a low density residential subwatershed (IC < 5%), as much as 75% of the annual runoff volume could occur as baseflow. In such a case, annual baseflow load may be equivalent to the annual stormflow load.

4.4 Sediment

Sediment is an important and ubiquitous pollutant in urban stormwater runoff. Sediment can be measured in three distinct ways: Total Suspended Solids (TSS), Total Dissolved Solids (TDS) and turbidity. TSS is a measure of the total mass suspended sediment particles in water. The measurement of TSS in urban stormwater helps to estimate sediment load transported to local and downstream receiving waters. Table 22 summarizes stormwater EMCs for total suspended solids, as reported by Barrett *et al.* (1995), Smullen and Cave (1998), and USEPA (1983). TDS is a measure of the dissolved solids and minerals present in stormwater runoff and is used as a primary indication of the purity of drinking water. Since few stormwater monitoring efforts have focused on TDS, they are not reported in this document. Turbidity is a measure of how suspended solids present in water reduce the ability of light to penetrate the water column. Turbidity can exert impacts on aquatic biota, such as the ability of submerged aquatic vegetation to receive light and the ability of fish and aquatic insects to use their gills (Table 23).

4.4.1 Concentrations

TSS concentrations in stormwater across the country are well documented. Table 18 reviews mean TSS EMCs from 13 communities across the country and reveals a wide range of recorded concentrations. The lowest concentration of 43 mg/l was reported in Florida, while TSS reached 663 mg/l in Dallas, Texas. Variation in sediment concentrations has been attributed to regional rainfall differences (Driver, 1988); construction site runoff (Leopold, 1968); and bank erosion (Dartiguenave *et al.*, 1997). National values are provided in Table 22.

Turbidity levels are not as frequently reported in national and regional monitoring summaries. Barrett and Malina (1998) monitored turbidity at two sites in Austin, Texas and reported a mean turbidity of 53 NTU over 34 storm events (Table 22).

4.4.2 Impacts of Sediment on Streams

The impacts of sediment on aquatic biota are well documented and can be divided into impacts caused by suspended sediment and those caused by deposited sediments (Tables 23 and 24).

In general, high levels of TSS and/or turbidity can affect stream habitat and cause sedimentation in downstream receiving waters. Deposited sediment can cover benthic organisms such as aquatic insects and freshwater mussels. Other problems associated with high sediments loads include stream warming by reflecting radiant energy due to increased turbidity (Kundell and Rasmussen, 1995), decreased flow capacity (Leopold, 1973), and increasing overbank flows (Barrett and Malina, 1998). Sediments also transport other pollutants which bind to sediment particles. Significant levels of pollutants can be transported by sediment during stormwater runoff events,

Table 22: EMCs for Total Suspended Solids and Turbidity

Pollutant	EMCs		Number of Events	Source
	Mean	Median		
TSS (mg/l)	78.4	54.5	3047	Smullen and Cave, 1998
	174	113	2000	USEPA, 1983
Turbidity (NTU)	53	N/R	423	Barrett and Malina, 1998

N/R = Not Reported

Table 23: Summary of Impacts of Suspended Sediment on the Aquatic Environment (Schueler and Holland, 2000)
Abrades and damages fish gills, increasing risk of infection and disease
Scouring of periphyton from stream (plants attached to rocks)
Loss of sensitive or threatened fish species when turbidity exceeds 25 NTU
Shifts in fish community toward more sediment-tolerant species
Decline in sunfish, bass, chub and catfish when month turbidity exceeds 100 NTU
Reduces sight distance for trout, with reduction in feeding efficiency
Reduces light penetration causing reduction in plankton and aquatic plant growth
Adversely impacts aquatic insects, which are the base of the food chain
Slightly increases the stream temperature in the summer
Suspended sediments can be a major carrier of nutrients and metals
Reduces anglers chances of catching fish

Table 24: Summary of Impacts of Deposited Sediments on the Aquatic Environment (Schueler and Holland, 2000)
1. Physical smothering of benthic aquatic insect community
2. Reduced survival rates for fish eggs
3. Destruction of fish spawning areas and eggs
4. Embeddedness of stream bottom reduced fish and macroinvertebrate habitat value
5. Loss of trout habitat when fine sediments are deposited in spawning or riffle-runs
6. Sensitive or threatened darters and dace may be eliminated from fish community
7. Increase in sediment oxygen demand can deplete dissolved oxygen in streams
8. Significant contributing factor in the alarming decline of freshwater mussels
9. Reduced channel capacity, exacerbating downstream bank erosion and flooding
10. Reduced flood transport capacity under bridges and through culverts
11. Deposits diminish scenic and recreational values of waterways

including trace metals, hydrocarbons and nutrients (Crunkilton *et al.,* 1996; Dartiguenave *et al*., 1997; Gavin and Moore, 1982; Novotny and Chester, 1989; Schueler 1994b).

4.4.3 Sources and Source Areas of Sediment

Sediment sources in urban watersheds include stream bank erosion; erosion from exposed soils, such as from construction sites; and washoff from impervious areas (Table 25).

As noted in this chapter, streambank erosion is generally considered to be the primary source of sediment to urban streams. Recent studies by Dartiguenave *et al.* (1997) and Trimble (1997) determined that streambank erosion contributes the majority of the annual sediment budget of urban streams. Trimble (1997) directly measured stream cross sections, sediment aggradation and suspended sediment loads and determined that two-thirds of the annual sediment budget of a San Diego, California watershed was supplied by streambank erosion. Dartiguenave *et al.* (1997) developed a GIS based model in Austin, Texas to determine the effects of stream bank erosion on the annual sediment budget. They compared modeled sediment loads from the watershed with the actual sediment loads measured at USGS gaging stations and concluded that more than 75% of the sediment load came from streambank erosion. Dartiguenave *et al.* (1997) reported that sediment load per unit area increases with increasing IC (Figure 31).

Sediment loads are also produced by washoff of sediment particles from impervious areas and their subsequent transport in stormwater runoff sediment. Source areas include parking lots, streets, rooftops, driveways and lawns. Streets and parking lots build up dirt and grime from the wearing of the street surface, exhaust particulates, "blown on" soil and organic matter, and atmospheric deposition. Lawn runoff primarily contains soil and organic matter. Urban source areas that produce the highest TSS concentrations include streets, parking lots and lawns (Table 26).

Parking lots and streets are not only responsible for high concentrations of sediment but also high runoff volumes. The SLAMM source loading model (Pitt and Voorhees, 1989) looks at runoff volume and concentrations of pollutants from different urban land uses and predicts stream loading. When used in the Wisconsin and Michigan subwatersheds, it demonstrated that parking lots and streets were responsible for over 70% of the TSS delivered to the stream. (Steuer *et al*., 1997; Waschbusch *et al*., 2000).

Table 25: Sources and Loading of Suspended Solids Sediment in Urban Areas

Sources	Loading	Source
Bank Erosion	75% of stream sediment budget	Dartinguenave *et al*., 1997
	66% of stream sediment budget	Trimble, 1997
Overland Flow- Lawns	397 mg/l (geometric mean)	Bannerman *et al*., 1993
	262 mg/l	Steuer *et al*., 1997
	11.5% (estimated; 2 sites)	Waschbusch *et al*., 2000
Construction Sites	200 to 1200 mg/l	Table 27
Washoff from Impervious Surfaces	78 mg/l (mean)	Table 16

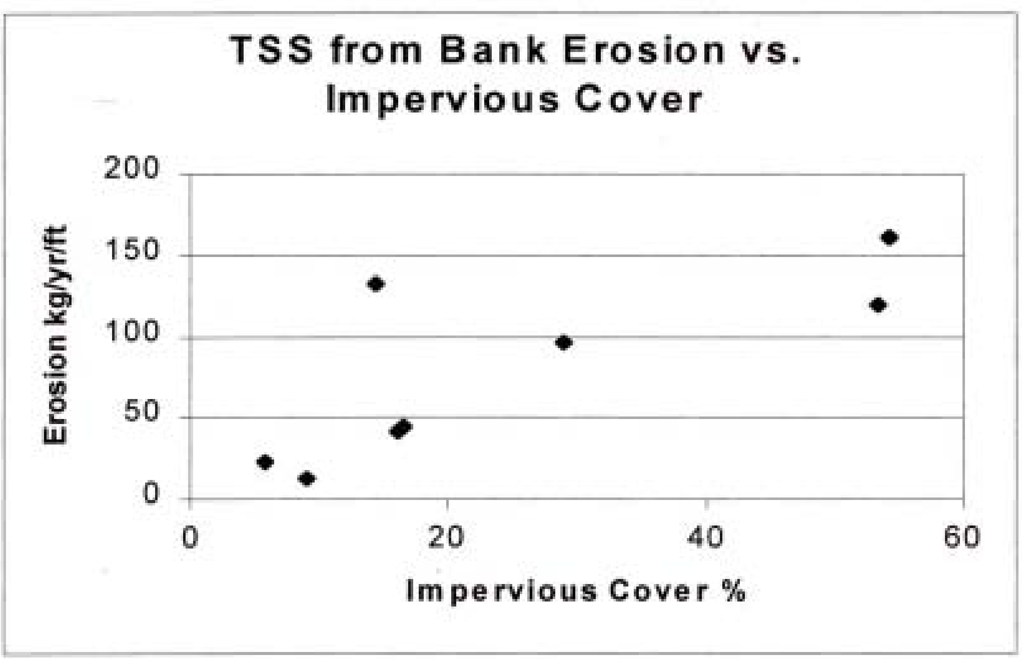


Figure 31: TSS from Bank Erosion vs. IC in Texas Streams (Daringuenave *et al*., 1997)

The third major source of sediment loads is erosion from construction sites. Several studies have reported extremely high TSS concentrations in construction site runoff, and these findings are summarized in Table 27. TSS concentrations from uncontrolled construction sites can be more than 150 times greater than those from undeveloped land (Leopold, 1968) and can be reduced if erosion and sediment control practices are applied to construction sites.

Table 26: Source Area Geometric Mean Concentrations for Suspended Solids in Urban Areas

Source Area	Suspended Solids (mg/l)		
Source	(1)	(2)	(3)
Commercial Parking Lot	110	58	51
High Traffic Street	226	232	65
Medium Traffic Street	305	326	51
Low Traffic Street	175	662	68
Commercial Rooftop	24	15	18
Residential Rooftop	36	27	15
Residential Driveway	157	173	N/R
Residential Lawn	262	397	59

Sources: (1) Steuer et al., 1997; (2) Bannerman et al., 1993; (3) Waschbusch et al., 2000; N/R = Not Reported

Table 27: Mean TSS Inflow and Outflow at Uncontrolled, Controlled and Simulated Construction Sites

Source	Mean Inflow TSS Concentration (mg/l)	Mean Outflow TSS Concentration (mg/l)	Location
Uncontrolled Sites			
Horner *et al.*, 1990	7,363	281	PNW
Schueler and Lugbill,1990	3,646	501	MD
York and Herb, 1978	4,200	N/R	MD
Islam *et al.*, 1988	2,950	N/R	OH
Controlled Sites			
Schueler and Lugbill, 1990	466	212	MD
Simulated Sediment Concentrations			
Jarrett, 1996	9,700	800	PA
Sturm and Kirby, 1991	1,500-4,500	200-1,000	GA
Barfield and Clar, 1985	1,000-5,000	200-1,200	MD
Dartiguenave *et al.*, 1997	N/R	600	TX

N/R = Not Reported

4.5 Nutrients

Nitrogen and phosphorus are essential nutrients for aquatic systems. However, when they appear in excess concentrations, they can exert a negative impact on receiving waters. Nutrient concentrations are reported in several ways. Nitrogen is often reported as nitrate (NO_3) and nitrite (NO_2), which are inorganic forms of nitrogen; total nitrogen (Total N), which is the sum of nitrate, nitrite, organic nitrogen and ammonia; and total Kjeldhal nitrogen (TKN), which is organic nitrogen plus ammonia.

Phosphates are frequently reported as soluble phosphorus, which is the dissolved and reactive form of phosphorus that is available for uptake by plants and animals. Total phosphorus (Total P) is also measured, which includes both organic and inorganic forms of phosphorus. Organic phosphorus is derived from living plants and animals, while inorganic phosphate is comprised of phosphate ions that are often bound to sediments.

4.5.1 Concentrations

Many studies have indicated that nutrient concentrations are linked to land use type, with urban and agricultural watersheds producing the highest nutrient loads (Chessman *et al.* 1992; Paul *et al.*, 2001; USGS, 2001b and Wernick *et al.*,1998). Typical nitrogen and phosphorus EMC data in urban stormwater runoff are summarized in Table 28.

Some indication of the typical concentrations of nitrate and phosphorus in stormwater runoff are evident in Figures 32 and 33. These graphs profile average EMCs in stormwater runoff recorded at 37 residential catchments across the U.S. The average nitrate EMC is remarkably consistent among residential neighborhoods, with most clustered around the mean of 0.6 mg/l and a range of 0.25 to 1.4 mg/l. The concentration of phosphorus during storms is also very consistent with a mean of 0.30 mg/l and a rather tight range of 0.1 to 0.66 mg/l (Schueler, 1995).

The amount of annual rainfall can also influence the magnitude of nutrient concentrations in stormwater runoff. For example, both Caraco (2000a) and Driver (1988) reported that the highest nutrient EMCs were found in stormwater from arid or semi-arid regions.

Table 28: EMCs of Phosphorus and Nitrogen Urban Stormwater Pollutants

Pollutant	EMCs (mg/l)		Number of Events	Source
	Mean	Median		
Total P	0.315	0.259	3094	Smullen and Cave, 1998
	0.337	0.266	1902	USEPA, 1983
Soluble P	0.129	0.103	1091	Smullen and Cave, 1998
	0.1	0.078	767	USEPA, 1983
Total N	2.39	2.00	2016	Smullen and Cave, 1998
	2.51	2.08	1234	USEPA, 1983
TKN	1.73	1.47	2693	Smullen and Cave, 1998
	1.67	1.41	1601	USEPA, 1983
Nitrite & Nitrate	0.658	0.533	2016	Smullen and Cave, 1998
	0.837	0.666	1234	USEPA, 1983

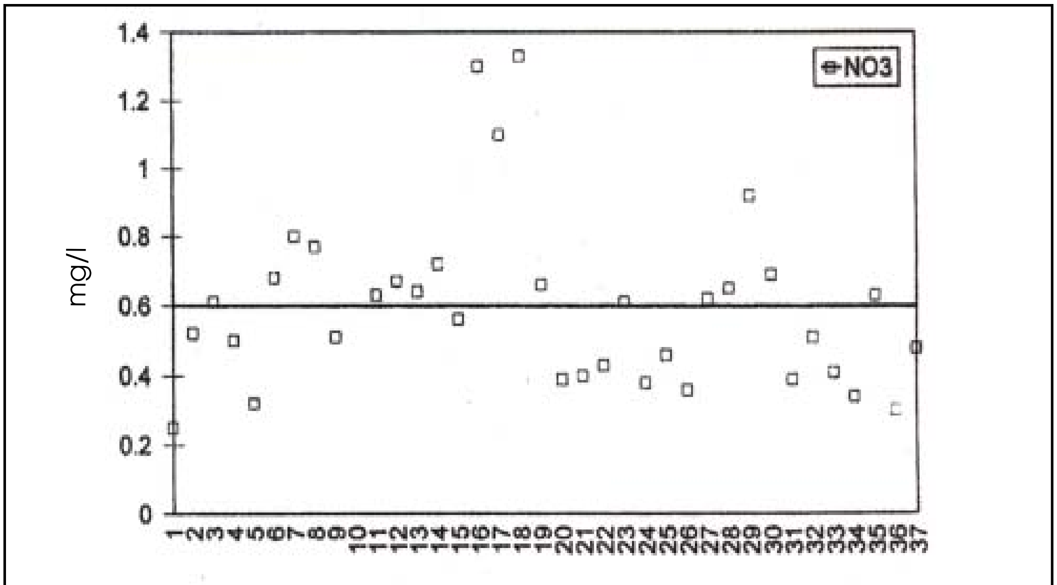


Figure 32: Nitrate-Nitrogen Concentration in Stormwater Runoff at 37 Sites Nationally (Schueler, 1999)

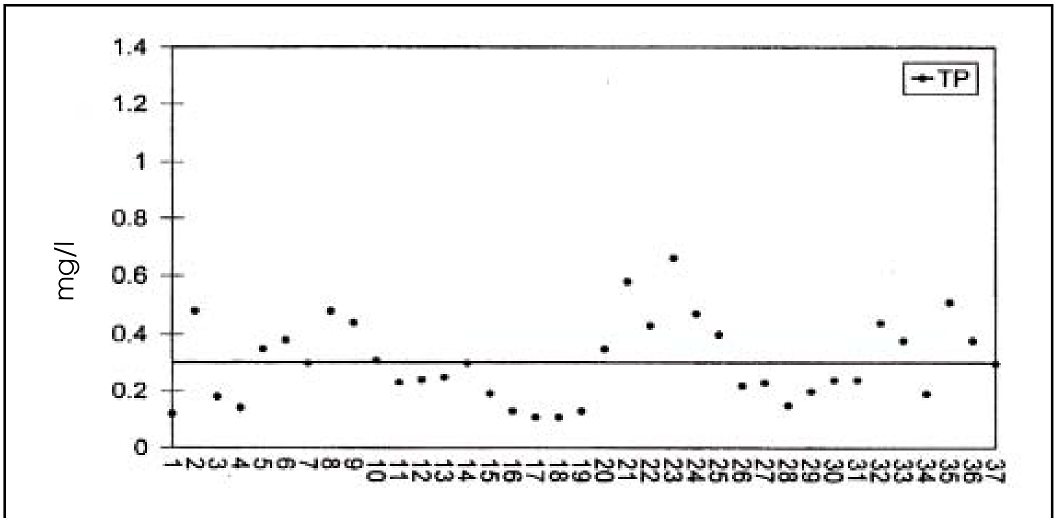


Figure 33: Total Phosphorus Concentration in Stormwater at 37 Sites Nationally (Schueler, 1999)

4.5.2 Impacts of Nutrients on Streams

Much research on the impact of nutrient loads has been focused on lakes, reservoirs and estuaries, which can experience eutrophication. Nitrogen and phosphorus can contribute to algae growth and eutrophic conditions, depending on which nutrient limits growth (USEPA, 1998). Dissolved oxygen is also affected by eutrophication. When algae or aquatic plants that are stimulated by excess nutrients die off, they are broken down by bacteria, which depletes the oxygen in the water. Relatively few studies have specifically explored the impact of nutrient enrichment on urban streams. Chessman *et al*. (1992) studied the limiting nutrients for periphyton growth in a variety of streams and noted that the severity of eutrophication was related to low flow conditions. Higher flow rates in streams may cycle nutrients faster than in slow flow rates, thus diminishing the extent of stream eutrophication.

4.5.3 Sources and Source Areas of Nutrients

Phosphorus is normally transported in surface water attached to sediment particles or in soluble forms. Nitrogen is normally transported by surface water runoff in urban watersheds. Sources for nitrogen and phosphorus in urban stormwater include fertilizer, pet waste, organic matter (such as leaves and detritus), and stream bank erosion. Another significant source of nutrients is atmospheric deposition. Fossil fuel combustion by automobiles, power plants and industry can supply nutrients in both wet fall and dry fall. The Metropolitan Washington Council of Governments (MWCOG, 1983) estimated total annual atmospheric deposition rates of 17 lbs/ac for nitrogen and 0.7 lbs/ac for phosphorus in the Washington, D.C. metro area.

Research from the upper Midwest suggests "hot spot" sources can exist for both nitrogen and phosphorus in urban watersheds. Lawns, in particular, contribute greater concentrations of Total N, Total P and dissolved phosphorus than other urban source areas. Indeed, source research suggests that nutrient concentrations in lawn runoff can be as much as four times greater than other urban sources such as streets, rooftops or driveways (Bannerman *et al.*, 1993; Steuer *et al.*, 1997 and Waschbusch *et al.*, 2000) (Table 29). This finding is significant, since lawns can comprise more than 50% of the total area in suburban watersheds. Lawn care, however, has seldom been directly linked to elevated nutrient concentrations during storms. A very recent lakeshore study noted that phosphorus concentrations were higher in fertilized lawns compared to unfertilized lawns, but no significant difference was noted for nitrogen (Garn, 2002).

Wash-off of deposited nutrients from IC is thought to be a major source of nitrogen and phosphorus during storms (MWCOG, 1983). While the concentration of nitrogen and phosphorus from parking lots and streets is lower than lawns, the volume of runoff is significantly higher. In two studies using the SLAMM source loading model (Pitt and Voorhees, 1989), parking lots and streets were responsible for over 30% of the nitrogen and were second behind lawns in their contributions to the phosphorus load (Steuer *et al.*, 1997; Waschbusch *et al.*, 2000).

Table 29: Source Area Monitoring Data for Total Nitrogen and Total Phosphorous in Urban Areas

Source Area	Total N (mg/l)	Total P (mg/l)		
Source	(1)	(1)	(2)	(3)
Commercial Parking Lot	1.94	0.20	N/R	0.10
High Traffic Street	2.95	0.31	0.47	0.18
Med. Traffic Street	1.62	0.23	1.07	0.22
Low Traffic Street	1.17	0.14	1.31	0.40
Commercial Rooftop	2.09	0.09	0.20	0.13
Residential Rooftop	1.46	0.06	0.15	0.07
Residential Driveway	2.10	0.35	1.16	N/R
Residential Lawn	9.70	2.33	2.67	0.79
Basin Outlet	1.87	0.29	0.66	N/R

(1) Steuer et al., 1997; (2) Bannerman et al., 1993; (3) Waschbusch et al., 2000; N/R= Not Reported

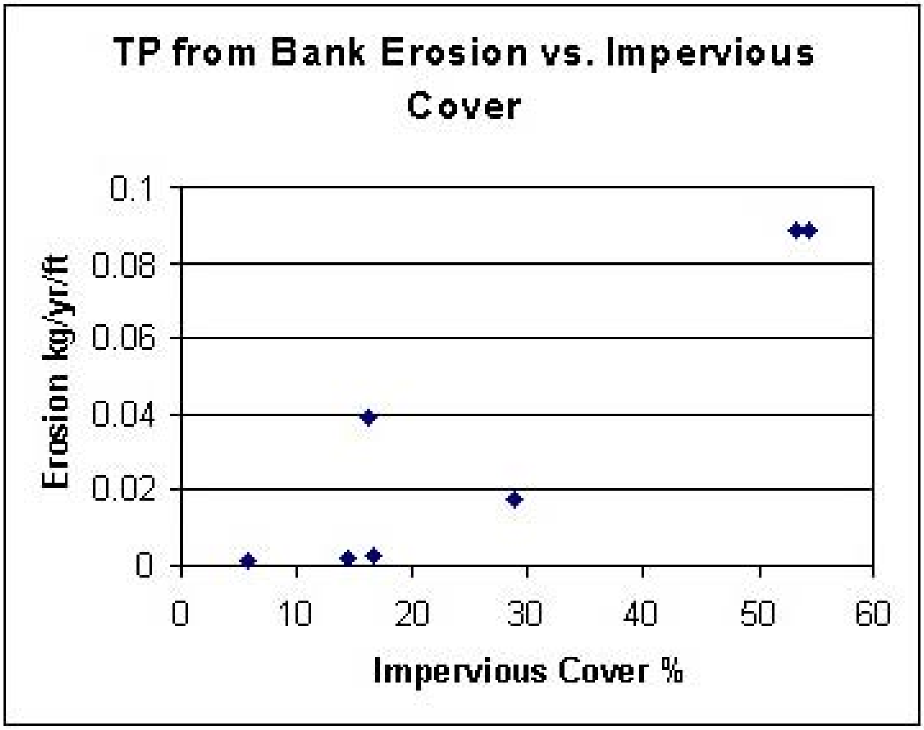


Figure 34: Total Phosphorus from Bank Erosion as a Function of IC in Texas Streams (Dartiguenave *et al.*, 1997)

Streambank erosion also appears to be a major source of nitrogen and phosphorus in urban streams. Both nitrogen and phosphorus are often attached to eroded bank sediment, as indicated in a recent study by Dartiguenave *et al.* (1997) in Austin, Texas. They showed that channel erosion contributed nearly 50% of the Total P load shown for subwatersheds with IC levels between 10 and 60 % (Figure 34). These findings suggest that prevention or reduction of downstream channel erosion may be an important nutrient reduction strategy for urban watersheds.

Snowmelt runoff generally has higher nutrient EMCs, compared to stormwater runoff. Oberts (1994) found that TKN and nitrate EMCs were much higher in snowmelt at all sites. The same pattern has also been observed for phosphorus EMCs during snowmelt and stormwater runoff. Zapf-Gilje *et al*. (1986) found that the first 20% of snowmelt events contained 65% of the phosphorus and 90% of the nitrogen load. Ayers *et al.* (1985) reported that a higher percentage of the annual nitrate, TKN and phosphorus load was derived from snowmelt runoff compared to stormwater runoff in an urban Minnesota watershed, which presumably reflects the accumulation of nutrients in the snowpack during the winter.

Metal	Detection Frequency(1)	EMCs

Table 31: Average Total Recoverable and Dissolved Metals for 13 Stormwater Flows and Nine Baseflow Samples from Lincoln Creek in 1994 (Crunkilton *et al.*, 1996)

	Total Recoverable		Dissolved	
Metal (Fg/l)	**Storm Flow**	**Baseflow**	**Storm Flow**	**Baseflow**
Lead	35	3	1.7	1.2
Zinc	133	22	13	8
Copper	23	7	5	4
Cadmium	0.6	0.1	0.1	0.1

higher risk of exceeding trace metal concentration standards.

Crunkilton *et al.* (1996) measured recoverable and dissolved metals concentrations in Lincoln Creek, Wisconsin and found higher EMCs during storm events compared to baseflow periods (Table 31). They also found that total recoverable metal concentrations were almost always higher than the dissolved concentration (which is the more available form).

4.6.2 Impacts of Trace Metals on Streams

Although a great deal is known about the concentration of metals in urban stormwater, much less is known about their possible toxicity on aquatic biota. The primary concern related to the presence of trace metals in streams is their potential toxicity to aquatic organisms. High concentrations can lead to bioaccumulation of metals in plants and animals, possible chronic or acute toxicity, and contamination of sediments, which can affect bottom dwelling organisms (Masterson and Bannerman, 1994). Generally, trace metal concentrations found in urban stormwater are not high enough to cause acute toxicity (Field and Pitt, 1990). The cumulative accumulation of trace metal concentrations in bottom sediments and animal tissues are of greater concern. Some evidence exists for trace metal accumulation in bottom sediments of receiving waters and for bioaccumulation in aquatic species (Bay and Brown, 2000 and Livingston, 1996).

Relatively few studies have examined the chronic toxicity issue. Crunkilton *et al.* (1996) found that concentrations of lead, zinc and copper exceeded EPA's Chronic Toxicity Criteria more than 75% of the time in stormflow in stormwater samples for Lincoln Creek in Wisconsin. When exposed to storm and base flows in Lincoln Creek, *Ceriodaphnia dubia*, a common invertebrate test species, demonstrated significant mortality in extended flow-through tests. Around 30% mortality was recorded after seven days of exposure and 70% mortality was recorded after 14 days.

Crunkilton *et al.* (1996) also found that significant mortality in bullhead minnows occurred in only 14% of the tests by the end of 14 days, but mortality increased to 100% during exposures of 17 to 61 days (see Table 32). In a related study in the same watershed, Masterson and Bannerman (1994) determined that crayfish in Lincoln Creek had elevated levels of lead, cadmium, chromium and copper when compared to crayfish from a reference stream. The Lincoln Creek research provides limited evidence that prolonged exposure to trace metals in urban streams may result in significant toxicity.

Most toxicity research conducted on urban stormwater has tested for acute toxicity over a short period of time (two to seven days). Shorter term whole effluent toxicity protocols are generally limited to seven days (Crunkilton *et al.*, 1996). Research by Ellis (1986) reported delayed toxicity in urban streams. Field and Pitt (1990) demonstrated that pollutants deposited to the stream during storm events may take upwards of 10 to 14 days to exert influence. The research suggests that longer term in-situ and flow-through monitoring are needed to definitively answer the question whether metal levels in stormwater can be chronically toxic.

An additional concern is that trace metals co-occur with other pollutants found in urban stormwater, and it is not clear whether they interact to increase or decrease potential toxicity. Hall and Anderson (1988) investigated the toxicity and chemical composition of urban stormwater runoff in British Columbia and found that the interaction of pollutants changed the toxicity of some metals. In laboratory analysis with *Daphnia pulex*, an aquatic invertebrate, they found that the toxicity of iron was low and that its presence reduced the toxicity of other metals. On the other hand, the presence of lead increased the toxicity of copper and zinc.

Interaction with sediment also influences the impact of metals. Often, over half of the trace metals are attached to sediment (MWCOG, 1983). This effectively removes the metals from the water column and reduces the availability for biological uptake and subsequent bioaccumulation (Gavin and Moore, 1982 and OWML, 1983). However, metals accumulated in bottom sediment can then be resuspended during storms (Heaney and Huber, 1978). It is important to note that the toxic effect of metals can be altered when found in conjunction with other substances. For instance, the presence of chlorides can increase the toxicity of some metals. Both metals and chlorides are common pollutants in snowpacks (see section 4.2 for more snow melt information).

4.6.3 Sources and Source Areas of Trace Metals

Research conducted in the Santa Clara Valley of California suggests that cars can be the dominant loading source for many metals of concern, such as cadmium, chromium, copper, lead, mercury and zinc (EOA, Inc., 2001). Other sources are also important and include atmospheric deposition, rooftops and runoff from industrial and residential sites.

The sources and source areas for zinc, copper, lead, chromium and cadmium are listed in Table 33. Source areas for trace metals in the urban environment include streets, parking lots, snowpacks and rooftops. Copper is often found in higher concentrations on urban streets, because some vehicles have brake pads that contain copper. For example, the Santa Clara study estimated that 50% of the total copper load was due to brake pad wear (Woodward-Clyde, 1992). Sources of lead include atmospheric deposition and diesel fuel emissions, which frequently occur along rooftops

Table 32: Percentage of In-situ Flow-through Toxicity Tests Using Daphnia *magna* and Pimephales *promelas* with Significant Toxic Effects from Lincoln Creek (Crunkilton *et al.*, 1996)

Species	Effect	Percent of Tests with Significant ($p<0.05$) Toxic Effects as Compared to Controls According to Exposure				
		48 hours	96 hours	7 days	14 days	17-61 days
D. *magna*	Mortality	0	N/R	36%	93%	N/R
	Reduced Reproduction	0	N/R	36%	93%	N/R
P. *promelas*	Mortality	N/R	0	0	14%	100%
	Reduced Biomass	N/R	N/R	60%	75%	N/R

N/R = Not Reported

and streets. Zinc in urban environments is a result of the wear of automobile tires (estimated 60% in the Santa Clara study), paints, and weathering of galvanized gutters and downspouts. Source area concentrations of trace metals are presented in Table 34. In general, trace metal concentrations vary considerably, but the relative rank among source areas remains relatively constant. For example, a source loading model developed for an urban watershed in Michigan estimated that parking lots, driveways and residential streets were the primary source areas for zinc, copper and cadmium loads (Steuer *et al.*, 1997).

Table 33: Metal Sources and Source Area "Hotspots" in Urban Areas

Metal	Sources	Source Area Hotspots
Zinc	tires, fuel combustion, galvanized pipes, roofs and gutters, road salts **estimate of 60% from tires*	parking lots, commercial and industrial rooftops, and streets
Copper	auto brake linings, pipes and fittings, algacides, and electroplating **estimate of 50% from brake pad wear*	parking lots, commercial roofs and streets
Lead	diesel fuel, paints and stains	parking lots, rooftops, and streets
Cadmium	component of motor oil and corrodes from alloys and plated surfaces	parking lots, rooftops, and streets
Chromium	found in exterior paints and corrodes from alloys and plated surfaces	most frequently found in industrial and commercial runoff

Sources: Bannerman et al., 1993; Barr, 1997; Steuer et al., 1997; Good, 1993; Woodward - Clyde, 1992

Table 34: Metal Source Area Concentrations in the Urban Landscape (*F*g/l)

Source Area	Dissolved Zinc	Total Zinc	Dissolved Copper		Total Copper	Dissolved Lead		Total Lead		
Source	(1)	(2)	(1)	(2)	(2)	(1)	(3)	(1)	(3)	(2)
Commercial Parking Lot	64	178	10.7	9	15	N/R	N/R	40	N/R	22
High Traffic Street	73	508	11.2	18	46	2.1	1.7	37	25	50
Medium Traffic Street	44	339	7.3	24	56	1.5	1.9	29	46	55
Low Traffic Street	24	220	7.5	9	24	1.5	.5	21	10	33
Commercial Rooftop	263	330	17.8	6	9	20	N/R	48	N/R	9
Residential Rooftop	188	149	6.6	10	15	4.4	N/R	25	N/R	21
Residential Driveway	27	107	11.8	9	17	2.3	N/R	52	N/R	17
Residential Lawn	N/R	59	N/R	13	13	N/R	N/R	N/R	N/R	N/R
Basin Outlet	23	203	7.0	5	16	2.4	N/R	49	N/R	32

Sources: (1) Steuer et al., 1997; (2) Bannerman et al., 1993; (3) Waschbusch, 2000; N/R = Not Reported

4.7 Hydrocarbons: PAH, Oil and Grease

Hydrocarbons are petroleum-based substances and are found frequently in urban stormwater. The term "hydrocarbons" is used to refer to measurements of oil and grease and polycyclic-aromatic hydrocarbons (PAH). Certain components of hydrocarbons, such as pyrene and benzo[b]fluoranthene, are carcinogens and may be toxic to biota (Menzie-Cura , 1995). Hydrocarbons normally travel attached to sediment or organic carbon. Like many pollutants, hydrocarbons accumulate in bottom sediments of receiving waters, such as urban lakes and estuaries. Relatively few studies have directly researched the impact of hydrocarbons on streams.

4.7.1 Concentrations

Table 35 summarizes reported EMCs of PAH and oil and grease derived from storm event monitoring at three different areas of the U.S. The limited research on oil and grease concentrations in urban runoff indicated that the highest concentrations were consistently found in commercial areas, while the lowest were found in residential areas.

4.7.2 Impacts of Hydrocarbons on Streams

The primary concern of PAH and oil and grease on streams is their potential bioaccumulation and toxicity in aquatic organisms. Bioaccumulation in crayfish, clams and fish has been reported by Masterson and Bannerman (1994); Moring and Rose (1997); and Velinsky and Cummins (1994).

Table 35: Hydrocarbon EMCs in Urban Areas

Hydrocarbon Indicator	EMC	Number of Events	Source	Location
	Mean			
PAH (Fg/l)	3.2*	12	Menzie-Cura, 1995	MA
	7.1	19	Menzie-Cura, 1995	MA
	13.4	N/R	Crunkilton *et al.*, 1996	WI
Oil and Grease (mg/l)	1.7 **R**** 9 **C** 3 **I**	30	Baird *et al.*, 1996	TX
	3	N/R	USEPA, 1983	U.S.
	5.4*	8	Menzie-Cura, 1995	MA
	3.5	10	Menzie-Cura, 1995	MA
	3.89 **R** 13.13 **C** 7.10 **I**	N/R	Silverman *et al.*, 1988	CA
	2.35 **R** 5.63 **C** 4.86 **I**	107	Barr, 1997	MD

*N/R = Not Reported; R = Residential, C = Commercial, I = Industrial; * = geometric mean, ** = median*

Moring and Rose (1997) also showed that not all PAH compounds accumulate equally in urban streams. They detected 24 different PAH compounds in semi-permeable membrane devices (SPMDs), but only three PAH compounds were detected in freshwater clam tissue. In addition, PAH levels in the SPMDs were significantly higher than those reported in the clams.

While acute PAH toxicity has been reported at extremely high concentrations (Ireland *et al.*, 1996), delayed toxicity has also been found (Ellis, 1986). Crayfish from Lincoln Creek had a PAH concentration of 360 Fg/kg, much higher than the concentration thought to be carcinogenic (Masterson and Bannerman, 1994). By comparison, crayfish in a non-urban stream had undetectable PAH levels. Toxic effects from PAH compounds may be limited since many are attached to sediment and may be less available, with further reduction occurring through photodegradation (Ireland *et al.,* 1996).

The metabolic effect of PAH compounds on aquatic life is unclear. Crunkilton *et al.* (1996) found potential metabolic costs to organisms, but Masterson and Bannerman (1994) and MacCoy and Black (1998) did not. The long-term effect of PAH compounds in sediments of receiving waters remains a question for further study.

4.7.3 Sources and Source Areas of Hydrocarbons

In most residential stormwater runoff, hydrocarbon concentrations are generally less than 5mg/l, but the concentrations can increase to five to 10 mg/l within some commercial, industrial and highway areas (See Table 35). Specific "hotspots" for hydrocarbons include gas stations, commuter parking lots, convenience stores, residential parking areas and streets (Schueler and Shepp, 1993). These authors evaluated hydrocarbon concentrations within oil and grease separators in the Washington Metropolitan area and determined that gas stations had significantly higher concentrations of hydrocarbons and trace metals, as compared to other urban source areas. Source area research in an urban catchment in Michigan showed that commercial parking lots contributed 64% of the total hydrocarbon load (Steuer *et al.*, 1997). In addition, highways were found to be a significant contributor of hydrocarbons by Lopes and Dionne (1998).

4.8 Bacteria and Pathogens

Bacteria are single celled organisms that are too small to see with the naked eye. Of particular interest are coliform bacteria, typically found within the digestive system of warm-blooded animals. The coliform family of bacteria includes fecal coliform, fecal streptococci and Escherichia coli, which are consistently found in urban stormwater runoff. Their presence confirms the existence of sewage or animal wastes in the water and indicates that other harmful bacteria, viruses or protozoans may be present, as well. Coliform bacteria are indicators of potential public health risks and not actual causes of disease.

A pathogen is a microbe that is actually known to cause disease under the right conditions. Two of the most common waterborne pathogens in the U.S. are the protozoans *Cryptosporidium parvum* and *Giardia lambia*. *Cryptosporidium* is a waterborne intestinal parasite that infects cattle and domestic animals and can be transmitted to humans, causing life-threatening problems in people with impaired immune systems (Xiao *et al.*, 2001). *Giardia* can cause intestinal problems in humans and animals when ingested (Bagley *et al.,* 1998). To infect new hosts, protozoans create hard casings known as oocysts (*Cryptosporidium*) or cysts (*Giardia*) that are shed in feces and travel through surface waters in search of a new host.

4.8.1 Concentrations

Concentrations of fecal coliform bacteria in urban stormwater typically exceed the 200 MPN/100 ml threshold set for human contact recreation (USGS, 2001b). Bacteria concentrations also tend to be highly variable from storm to storm. For example, a national summary of fecal coliform bacteria in stormwater runoff is shown in Figure 35 and Table 36. The variability in fecal coliform ranges from 10 to 500,000 MPN/100ml with a mean of 15,038 MPN/100ml (Schueler, 1999). Another national database of more than 1,600 stormwater events computed a mean concentration of 20,000

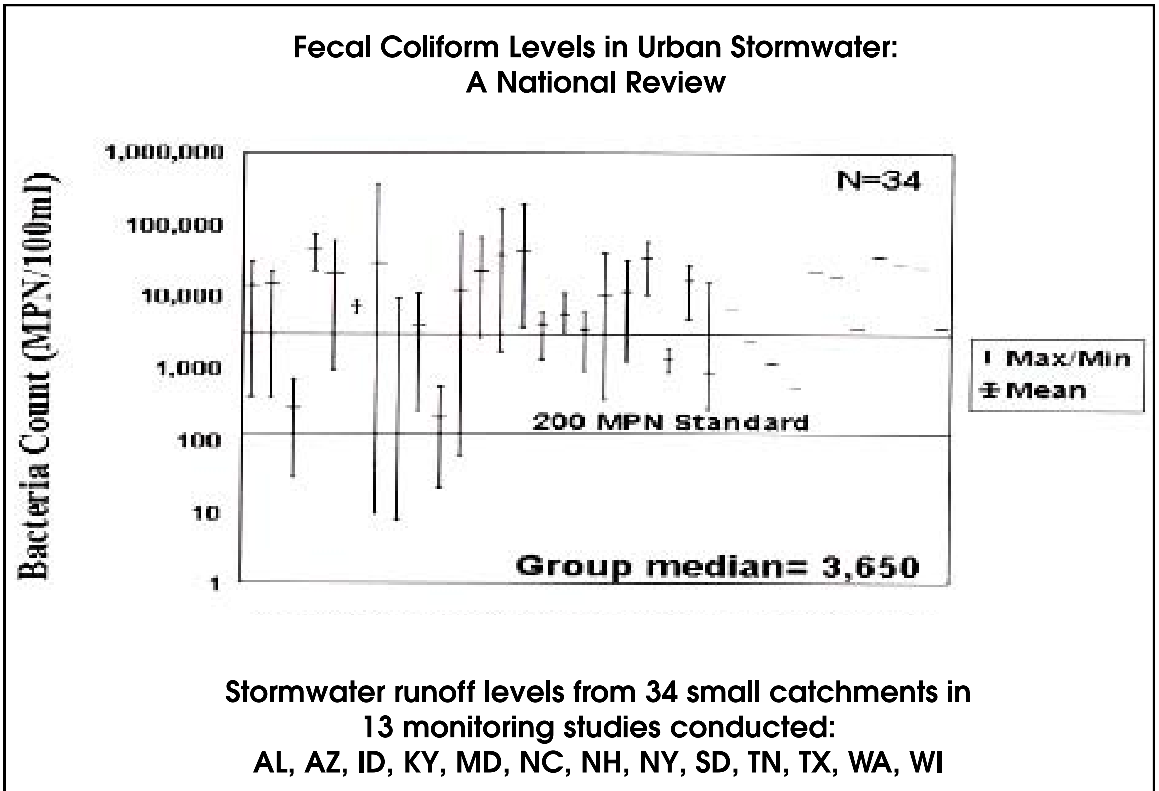


Figure 35: Fecal Coliform Levels in Urban Stormwater (Schueler, 1999)

Table 36: Bacteria EMCs in Urban Areas				
Bacteria Type	EMCs (MPN/100ml) Mean	Number of Events	Source	Location
Fecal Coliform	15,038	34	Schueler, 1999	U.S.
	20,000	1600	Pitt, 1998	U.S.
	7,653	27	Thomas and McClelland, 1995	GA
	20,000 **R*** 6900 **C** 9700 **I**	30*	Baird *et al.*, 1996	TX
	77,970	21 watersheds	Chang *et al.*, 1990	TX
	4,500	189	Varner, 1995	WA
	23,500	3	Young and Thackston, 1999	TN
Fecal Strep	35,351	17	Schueler, 1999	U.S.
	28,864 **R**	27	Thomas and McClelland, 1995	GA
	56,000 **R** * 18,000 **C** 6,100 **I**	30*	Baird *et al.*, 1996	TX

*N/R = Not Reported, R = Residential Area, C = Commercial Area, I = Industrial Area, * = Median*

MPN/100ml for fecal coliform (Pitt, 1998). Fecal streptococci concentrations for 17 urban sites across the country had a mean of 35,351 MPN/100ml (Schueler, 1999).

Young and Thackston (1999) showed that bacteria concentrations at four sites in metro Nashville were directly related to watershed IC. Increasing IC reflects the cumulative increase in potential bacteria sources in the urban landscape, such as failing septic systems, sewage overflows, dogs, and inappropriate discharges. Other studies show that concentrations of bacteria are typically higher in urban areas than rural areas (USGS, 1999a), but they are not always directly related to IC. For example, Hydroqual (1996) found that concentrations of fecal coliform in seven subwatersheds of the Kensico watershed in New York were generally higher for more developed basins, but fecal coliform concentrations did not directly increase with IC in the developed basins (Figure 36).

There is some evidence that higher concentrations of coliform are found in arid or semi-arid watersheds. Monitoring data from semi-arid regions in Austin, San Antonio, and Corpus Christi, Texas averaged 61,000, 37,500 and 40,500 MPN/100ml, respectively (Baird *et al.*,1996 and Chang *et al.* 1990). Schiff (1996), in a report of Southern California NPDES monitoring, found that median concentrations of fecal coliform in San Diego were 50,000 MPN/100ml and averaged 130,000 MPN/100ml in Los Angeles. In all of these arid and semi-arid regions, concentrations were significantly higher than the national average of 15,000 to 20,000 MPN/100ml.

Concentrations of *Cryptosporidium* and *Giardia* in urban stormwater are shown in Table 37. States *et al.* (1997) found high concentrations of *Cryptosporidium* and *Giardia* in storm samples from a combined sewer in Pittsburgh (geometric mean 2,013 oocysts/100ml and 28,881 cysts/100ml). There is evidence that urban stormwater runoff may have higher concentrations of *Cryptosporidium* and *Giardia* than other surface waters, as reported in Table 38 (Stern, 1996). Both pathogens were detected in about 50% of urban stormwater samples, suggesting some concern for drinking water supplies.

4.8.2 Impacts of Bacteria and Pathogens on Streams

Fecal coliform bacteria indicate the potential for harmful bacteria, viruses, or protozoans and are used by health authorities to determine public health risks. These standards were established to protect human health based on exposures to water during recreation and drinking. Bacteria standards for various water uses are presented in Table 39 and are all easily exceeded by typical urban stormwater concentrations. In fact, over 80,000 miles of streams and rivers are currently in non-attain-

Table 37: *Cryptosporidium* and *Giardia* EMCs

Pathogens	Units	EMCs		Number of Events	Source
		Mean	Median		
Cryptosporidium	oocysts	37.2	3.9	78	Stern, 1996
	oocysts/100ml	2013	N/R	N/R	States *et al.*, 1997
Giardia	cysts	41.0	6.4	78	Stern, 1996
	cysts/100ml	28,881	N/R	N/R	States *et al.*, 1997

N/R= Not reported

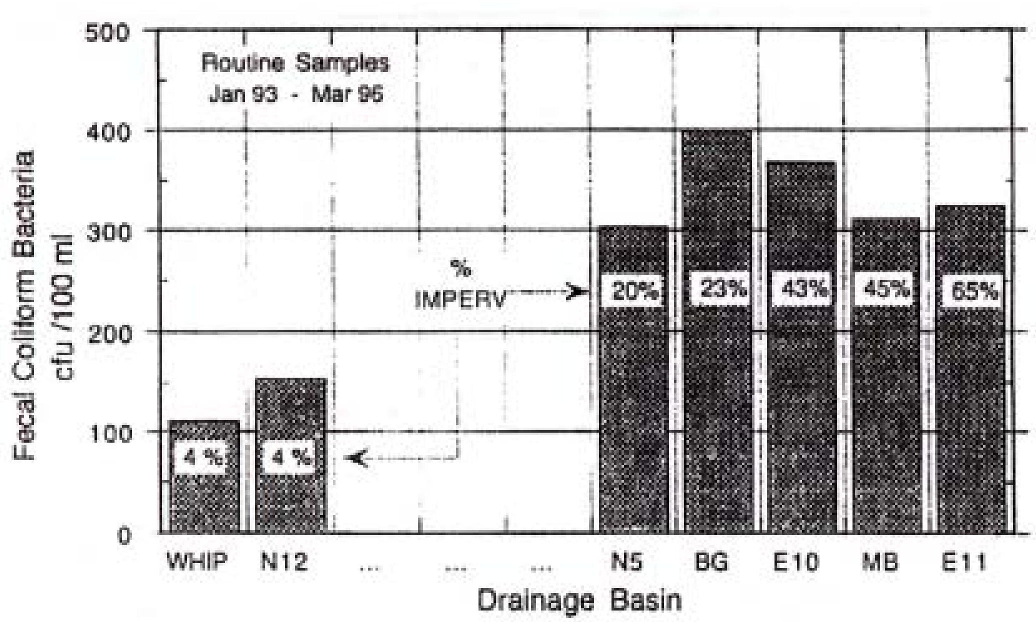


Figure 36: Relationship Between IC and Fecal Coliform Concentrations in New York Streams (Hydroqual, 1996)

Table 38: Percent Detection of *Giardia* cysts and *Cryptosporidium* oocysts in Subwatersheds and Wastewater Treatment Plant Effluent in the New York City Water Supply Watersheds (Stern, 1996)

Source Water Sampled	Number of Sources/ Number of Samples	Percent Detection			
		Total *Giardia*	Confirmed *Giardia*	Total *Cryptosporidium*	Confirmed *Cryptosporidium*
Wastewater Effluent	8/147	41.5%	12.9%	15.7%	5.4%
Urban Subwatershed	5/78	41.0%	6.4%	37.2%	3.9%
Agricultural Subwatershed	5/56	30.4%	3.6%	32.1%	3.6%
Undisturbed Subwatershed	5/73	26.0%	0.0%	9.6%	1.4%

Table 39: Typical Coliform Standards for Different Water Uses (USEPA, 1998)

Water Use	Microbial Indicator	Typical Water Standard
Water Contact Recreation	Fecal Coliform	<200 MPN per 100ml
Drinking Water Supply	Fecal Coliform	<20 MPN per 100ml
Shellfish Harvesting	Fecal Coliform	<14 MPN/ 100ml
Treated Drinking Water	Total Coliform	No more than 1% coliform positive samples per month
Freshwater Swimming	E.Coli	<126 MPN per 100ml

Important Note: Individual state standards may employ different sampling methods, indicators, averaging periods, averaging methods, instantaneous maximums and seasonal limits. MPN = most probable number. Higher or lower limits may be prescribed for different water use classes.

ment status because of high fecal coliform levels (USEPA, 1998).

4.8.3 Sources and Source Areas of Bacteria and Pathogens

Sources of coliform bacteria include waste from humans and wildlife, including livestock and pets. Essentially, any warm-blooded species that is present in significant numbers in a watershed is a potential culprit. Source identification studies, using methods such as DNA fingerprinting, have put the blame on species such as rats in urban areas, ducks and geese in stormwater ponds, livestock from hobby farms, dogs and even raccoons (Blankenship, 1996; Lim and Olivieri, 1982; Pitt, 1998; Samadpour and Checkowitz, 1998).

Transport of bacteria takes place through direct surface runoff, direct inputs to receiving waters, or indirect secondary sources. Source areas in the urban environment for direct runoff include lawns and turf, driveways, parking lots and streets. For example, dogs have high concentrations of fecal coliform in their feces and have a tendency to defecate in close proximity to IC (Schueler, 1999). Weiskel *et al.* (1996) found that direct inputs of fecal coliform from waterfowl can be very important; these inputs accounted for as much as 67% of the annual coliform load to Buttermilk Bay, Massachusetts.

Indirect sources of bacteria include leaking septic systems, illicit discharges, sanitary sewer overflows (SSOs), and combined sewer overflows (CSOs). These sources have the potential to deliver high coliform concentrations to urban streams. In fact, extremely high bacteria concentrations are usually associated with wastewater discharges. CSOs and SSOs occur when the flow into the sewer exceeds the capacity of the sewer lines to drain them. CSOs result from stormwater flow in the lines, and SSOs are a result of infiltration problems or blockages in the lines.

Illicit connections from businesses and homes to the storm drainage system can discharge sewage or washwater into receiving waters. Illicit discharges can often be identified by baseflow sampling of storm sewer systems. Leaking septic systems are estimated to comprise between 10 and 40% of the systems, and individual inspections are the best way to determine failing systems (Schueler, 1999).

There is also evidence that coliform bacteria can survive and reproduce in stream sediments and storm sewers (Schueler, 1999). During a storm event, they often become resuspended and add to the in-stream bacteria load. Source area studies reported that end of pipe concentrations were an order of magnitude higher than any source area on the land surface; therefore, it is likely that the storm sewer system itself acts as a source of fecal coliform (Bannerman *et al.*, 1993 and Steuer *et al.*, 1997). Resuspension of fecal coliform from fine stream sediments during storm events has been reported in New Mexico (NMSWQB, 1999). The sediments in-stream and in the storm sewer system may be significant contributors to the fecal coliform load.

Sources of *Cryptosporidium* and *Giardia* include human sewage and animal feces. *Cryptosporidium* is commonly found in cattle, dogs and geese. Graczyk *et al.* (1998) found that migrating Canada geese were a vector for *Cryptosporidium* and *Giardia*, which has implications for water quality in urban ponds that support large populations of geese.

4.9 Organic Carbon

Total organic carbon (TOC) is often used as an indicator of the amount of organic matter in a water sample. Typically, the more organic matter present in water, the more oxygen consumed, since oxygen is used by bacteria in the decomposition process. Adequate levels of dissolved oxygen in streams and receiving waters are important because they are critical to maintain aquatic life. Organic carbon is routinely found in urban stormwater, and high concentrations can result in an increase in Biochemical Oxygen Demand (BOD) and Chemical Oxygen Demand (COD). BOD and COD are measures of the oxygen demand caused by the decay of organic matter.

4.9.1 Concentrations

Urban stormwater has a significant ability to exert a high oxygen demand on a stream or receiving water, even two to three weeks after an individual storm event (Field and Pitt, 1990). Average concentrations of TOC, BOD and COD in urban stormwater are presented in Table 40. Mean concentrations of TOC, BOD and COD during storm events in nationwide studies were 17 mg/l, 14.1 mg/l and 52.8 mg/l, respectively (Kitchell, 2001 and Smullen and Cave,1998).

4.9.2 Impacts of Organic Carbon on Streams

TOC is primarily a concern for aquatic life because of its link to oxygen demand in streams, rivers, lakes and estuaries. The initial effect of increased concentrations of TOC, BOD or COD in stormwater runoff may be a depression in oxygen levels, which may persist for many days after a storm, as deposited organic matter gradually decomposes (Field and Pitt, 1990).

TOC is also a concern for drinking water quality. Organic carbon reacts with chlorine during the drinking water disinfection process and forms trihalomethanes and other disinfection by-products, which can be a serious drinking water quality problem (Water, 1999). TOC concentrations greater than 2 mg/l in treated water and 4 mg/l in source water can result in unacceptably high levels of disinfection byproducts and must be treated to reduce TOC or remove the disinfection byproducts (USEPA, 1998). TOC can also be a carrier for other pollutants, such as trace metals, hydrocarbons and nutrients.

4.9.3 Sources and Source Areas of Total Organic Carbon

The primary sources of TOC in urban areas appear to be decaying leaves and other organic matter, sediment and combustion by-products. Source areas include curbs, storm drains, streets and stream channels. Dartiguenave *et al.* (1997) determined that about half of the annual TOC load in urban watersheds of Austin, TX was derived from the eroding streambanks.

Table 40: EMCs for Organic Carbon in Urban Areas

Organic Carbon Source	EMCs (mg/l)		Number of Events	Source
	Mean	Median		
Total Organic Carbon (TOC)	32.0	N/R	423	Barrett and Malina, 1998
	17	15.2	19 studies	Kitchell, 2001
Biological Oxygen Demand (BOD)	14.1	11.5	1035	Smullen and Cave, 1998
	10.4	8.4	474	USEPA, 1983
Chemical Oxygen Demand (COD)	52.8	44.7	2639	Smullen and Cave, 1998
	66.1	55	1538	USEPA, 1983

N/R = Not Reported

4.10 MTBE

Methyl tertiary butyl-ether (MTBE) is a volatile organic compound (VOC) that is added to gasoline to increase oxygen levels, which helps gas burn cleaner (called an oxygenate). MTBE has been used as a performance fuel additive since the 1970s. In 1990, the use of oxygenates was mandated by federal law and concentrations of MTBE in gasoline increased. Today, MTBE is primarily used in large metropolitan areas that experience air pollution problems. Since 1990, MTBE has been detected at increasing levels in both surface water and groundwater and is one of the most frequently detected VOCs in urban watersheds (USGS, 2001a). EPA has declared MTBE to be a potential human carcinogen at high doses. In March 2000, a decision was made by EPA to follow California's lead to significantly reduce or eliminate the use of MTBE in gasoline.

4.10.1 Concentrations

MTBE is highly soluble in water and therefore not easily removed once it enters surface or ground water. Delzer (1999) detected the presence of MTBE in 27% of the shallow wells monitored in eight urban areas across the country (Figure 37). Detection frequency was significantly higher in New England and Denver, as shown in Table 41. In a second study conducted in 16 metropolitan areas, Delzer (1999) found that 83% of MTBE detections occurred between October and March, the time when MTBE is primarily used as a fuel additive. The median MTBE concentration was 1.5 ppb, well below EPA's draft advisory level of 20 ppb (Delzer, 1996).

4.10.2 Impacts of MTBE on Streams

The primary concerns regarding MTBE are that it is a known carcinogen to small mammals, a suspected human carcinogen at higher

Table 41: MTBE Detection Frequency

Location	Detection Frequency	Source	Year
211 shallow wells in eight urban areas	27%	Delzer	1999
Surface water samples in 16 metro areas	7%	Delzer	1996

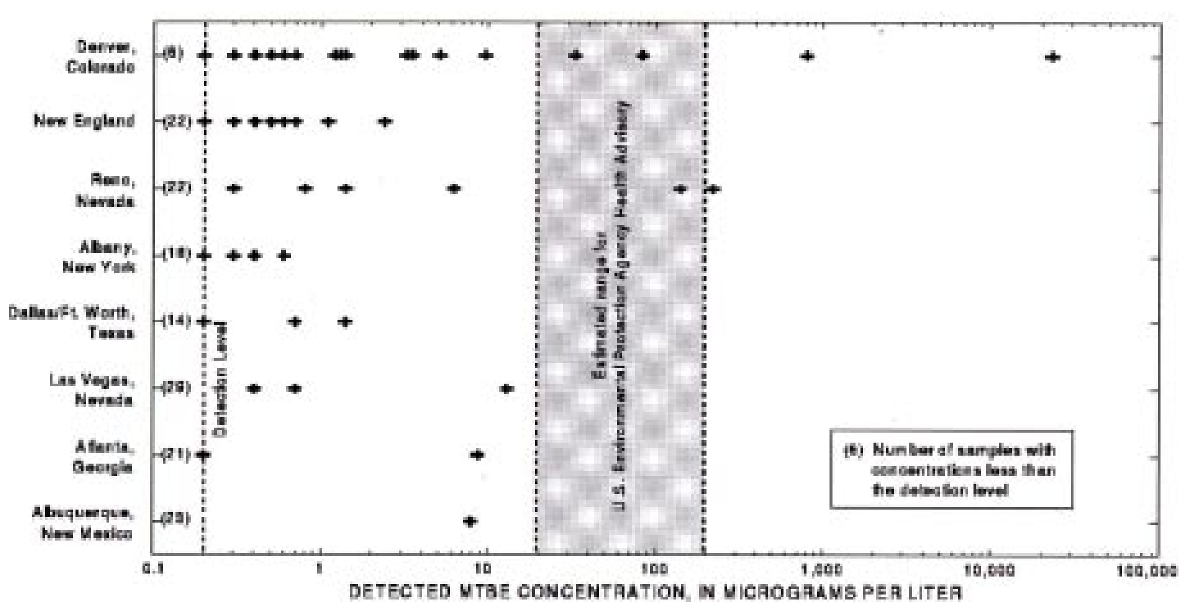


Figure 37: MTBE Concentrations in Surface Water from Eight Cities (Delzer, 1996)

doses and may possibly be toxic to aquatic life in small streams (Delzer, 1996). MTBE can also cause taste and odor problems in drinking water at fairly low concentrations. EPA issued a Drinking Water Advisory in 1997 that indicated that MTBE concentrations less than 20 ppb should not cause taste and odor problems for drinking water. However, the Association of California Water Agencies reports that some consumers can detect MTBE at levels as low as 2.5 ppb (ACWA, 2000). Because MTBE is frequently found in groundwater wells, it is thought to be a potential threat to drinking water (Delzer, 1999). For example, Santa Monica, California reportedly lost half of its groundwater drinking water supply due to MTBE contamination (Bay and Brown, 2000). MTBE has also been detected in human blood, especially in people frequently exposed to gasoline, such as gas station attendants (Squillace *et al.*, 1995).

4.10.3 Sources and Source Areas of MTBE

Since MTBE is a gasoline additive, its potential sources include any area that produces, transports, stores, or dispenses gasoline, particularly areas that are vulnerable to leaks and spills. Leaking underground storage tanks are usually associated with the highest MTBE concentrations in groundwater wells (Delzer, 1999). Vehicle emissions are also an important source of MTBE. Elevated levels are frequently observed along road corridors and drainage ditches. Once emitted, MTBE can travel in stormwater runoff or groundwater. Main source areas include heavily used multi-lane highways. Gas stations may also be a hotspot source area for MTBE contamination.

Another potential source of MTBE is watercraft, since two cycle engines can discharge as much as 20 to 30% of their fuel through the exhaust (Boughton and Lico, 1998). MTBE concentrations are clearly associated with increased use of gas engines, and there is concern that MTBE is an increasing component of atmospheric deposition (Boughton and Lico, 1998 and UC Davis, 1998).

4.11 Pesticides

Pesticides are used in the urban environment to control weeds, insects and other organisms that are considered pests. EPA estimates that nearly 70 million pounds of active pesticide ingredients are applied to urban lawns each year as herbicides or insecticides. Herbicides are used on urban lawns to target annual and perennial broadleaf weeds, while insecticides are used to control insects. Many types of pesticides are available for use in urban areas. Immerman and Drummond (1985) report that 338 different active ingredients are applied to lawns and gardens nationally. Each pesticide varies in mobility, persistence and potential aquatic impact. At high levels, many pesticides have been found to have adverse effects on ecological and human health. Several recent research studies by the USGS have shown that insecticides are detected with the greatest frequency in urban streams, and that pesticide detection frequency increases in proportion to the percentage of urban land in a watershed (Ferrari *et al.*, 1997; USGS, 1998, 1999a-b, 2001b). A national assessment by the USGS

Table 42: Median Concentrations and Detection Frequency of Herbicides and Insecticides in Urban Streams

Pollutant	Detection Frequency	Median Concentration (Fg/l)	Number of Samples	Source
Insecticides				
Diazinon	75%	0.025	326	USGS, 1998b
	92%	0.55	76	Brush *et al.*, 1995
	17%	0.002	1795	Ferrari *et al.*, 1997
Chlorpyrifos	41%	Non Detect	327	USGS, 1998b
	14%	0.004	1218	Brush *et al.*, 1995
Carbaryl	46%	Non Detect	327	USGS, 1998b
	22%	0.003	1128	Ferrari *et al.*, 1997
Herbicides				
Atrazine	86%	0.023	327	USGS, 1998b
	72%	0.099	2076	Ferrari *et al.*, 1997
Prometon	84%	0.031	327	USGS, 1998b
	56%	0.029	1531	Ferrari *et al.*, 1997
Simazine	88%	0.039	327	USGS, 1998b
	17%	0.046	1995	Ferrari *et al.*, 1997
2,4 -D	67%	1.1	11	Dindorf, 1992
	17%	0.035	786	Ferrari *et al.*, 1997
Dicamba	22%	1.8	4	Dindorf, 1992
MCPP	56%	1.8	10	Dindorf, 1992
MCPA	28%	1.0	5	Dindorf, 1992

(2001a) also indicates that insecticides are usually detected at higher concentrations in urban streams than in agricultural streams.

4.11.1 Concentrations

Median concentrations and detection frequency for common pesticides are shown in Table 42. Herbicides that are frequently detected in urban streams include atrazine; simazine; prometon; 2,4-D; dicamba; MCPP; and MCPA. Insecticides are also frequently encountered in urban streams, including diazinon, chlorpyrifos, malathion, and carbaryl. A USGS (1996) study monitored 16 sites in Gills Creek in Columbia, South Carolina over four days. This study reported that pesticide detection frequency increased as percent urban land increased.

Wotzka *et al.* (1994) monitored herbicide levels in an urban stream in Minneapolis, Minnesota during more than 40 storms. They found herbicides, such as 2,4-D; dicamba; MCPP; and MCPA in 85% of storm runoff events sampled. Total herbicide EMCs ranged from less than one to 70 µg/l. Ferrari *et al.* (1997) analyzed 463 streams in the mid-Atlantic region for the presence of 127 pesticide compounds. At least one pesticide was detected at more than 90% of the streams sampled.

Diazinon is one of the most commonly detected insecticides in urban stormwater runoff and dry weather flow. Diazinon was detected in 75% of National Water Quality Assessment (NAWQA) samples, 92% of stormflow samples from Texas, and 100% of urban stormflow samples in King County, Washington (Brush *et al.*, 1995 and USGS, 1999b). Diazinon is most frequently measured at concentrations greater than freshwater aquatic life criteria in urban stormwater (USGS, 1999a). USGS reports that diazinon concentrations were generally higher during urban stormflow (Ferrari *et al.,* 1997).

4.11.2 Impacts of Pesticides on Streams

Many pesticides are known or suspected carcinogens and can be toxic to humans and aquatic species. However, many of the known health effects require exposure to higher concentrations than typically found in the environment, while the health effects of chronic exposure to low levels are generally unknown (Ferrari *et al.*, 1997).

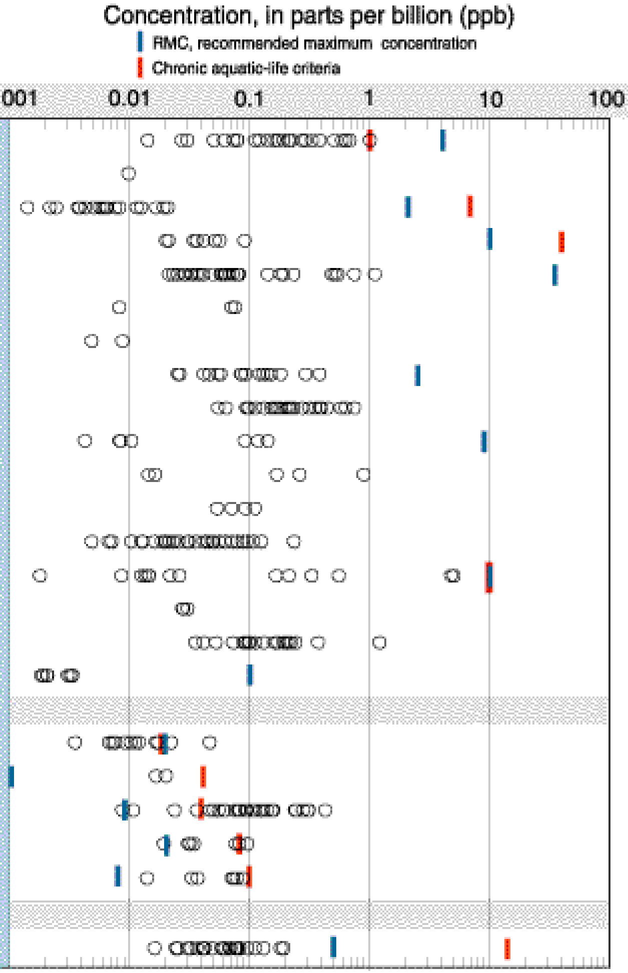
Studies that document the toxicity of insecticides and herbicides in urban stormwater have been focused largely on diazinon. Diazinon is responsible for the majority of acute toxicity in stormwater in Alameda County, California and King County, Washington (S.R. Hansen & Associates, 1995). Concentrations of diazinon in King County stormwater frequently exceed the freshwater aquatic life criteria (Figure 38). Similarly, research on Sacramento, California streams revealed acute toxicity for diazinon in 100% of stormwater samples using *Ceriodaphnia* as the test organism (Connor, 1995). Diazinon has a half-life of 42 days and is very soluble in water, which may explain its detection frequency and persistence in urban stormwater. Diazinon is also reported to attach fairly readily to organic carbon; consequently, it is likely re-suspended during storm events.

Insecticide concentrations exceeding acute and chronic toxicity thresholds for test organisms such as *Ceriodaphnia* have frequently been found in urban stormwater in New York, Texas, California, and Washington (Scanlin and Feng, 1997; Brush *et al.*, 1995; USGS, 1999b). The possibility exists that pesticides could have impacts on larger bodies of water, but there is a paucity of data on the subject at this time.

4.11.3 Sources and Source Areas of Pesticides

Sources for pesticides in urban areas include applications by homeowners, landscaping contractors and road maintenance crews. Source areas for pesticides in urban areas include lawns in residential areas; managed turf, such as golf courses, parks, and ball fields; and rights-of-way in nonresidential areas. Storage areas, which are subject to spills and leaks, can also be a source area. A study in San Francisco was able to trace high diazinon concentrations in some streams back to just a few households which had applied the pesticide at high levels (Scanlin and Feng, 1997). Two herbicides, simazine and atrazine, were detected in over 60% of samples in King County, WA stormwater but were not identified as being sold in retail stores. It is likely these herbicides are applied to nonresidential areas such as rights-of-way, parks and recreational areas (USGS, 1999b). Because pesticides are typically applied to turf, IC is not a direct indicator for pesticide concentrations, although they can drift onto paved surfaces and end up in stormwater runoff.

Active ingredient and trade name[1] example in parentheses	Freshwater aquatic-life criteria, in parts per billion (-- no aquatic-life criteria exist)		
Herbicide	**Acute**	**Chronic**	**RMC**
2,4-D (Weedone)	[4]10	[4]1	[5,2]4
Acetochlor (Guardian)	--	--	--
Atrazine (AAtrex)	[4]70	[4]7	[5]2
Dicamba (Banvel)	[4]390	[4]39	[5]10
Dichlobenil (Casoron)	--	--	[2]37
Dichlorprop (2,4-DP)	--	--	--
EPTC (Eptam)	--	--	--
MCPA (Kilsem)	--	--	[5]2.6
MCPP (Mecoprop)	--	--	--
Metolachlor (Dual)	--	--	[5]8
Napropamide (Devrinol)	--	--	--
Oxadiazon (Ronstar)	--	--	--
Prometon (Pramitol)	--	--	--
Simazine (Princep)	[4]100	[4]10	[2,5]10
Tebuthiuron (Spike)	--	--	--
Triclopyr (Garlon)	[4]5,600	[4]560	--
Trifluralin (Treflan)	--	--	[5]0.1
Insecticide			
Carbaryl (Sevin)	[4]0.17	[4]0.017	[2]0.02
Chlorpyrifos (Dursban)	[3,7]0.083	[3,7]0.041	[2]0.001
Diazinon (Diazinon)	[6]0.16	[6]0.04	[2]0.009
gamma-HCH (Lindane)	[3]0.95	[7]0.08	[2]0.02
Malathion (Malathion)	--	[3]0.1	[2]0.008
Fungicide			
Pentachlorophenol (Penta)	[3,8]19	[3,8]15	[5]0.5



[1] Use of trade names is for descriptive purposes and does not imply endorsement by the U.S. Geological Survey, the Department of Ecology, or King County
[2] National Academy of Sciences and National Academy of Engineering, 1973
[3] U.S. Environmental Protection Agency, 1998
[4] Norris and Dost, 1991, Table 3
[5] Ministers of Health Canada and Environment Canada, 1995
[6] Menconi and Cox, 1994
[7] State of Washington, 1992
[8] Criteria are pH dependent; values shown calculated at pH of 7.8

Figure 38: Concentrations of Pesticides in Stormwater in King County, WA (S.R. Hansen & Associates, 1995 and USGS, 1999b)

4.12 Deicers

Deicers are substances used to melt snow and ice to keep roads and walking areas safe. The most commonly used deicer is sodium chloride, although it may also be blended with calcium chloride or magnesium chloride. Other less frequently used deicers include urea and glycol, which are primarily used at airports to deice planes. Table 43 summarizes the composition, use and water quality effects of common deicers.

Chlorides are frequently found in snowmelt and stormwater runoff in most regions that experience snow and ice in the winter months (Oberts, 1994 and Sherman, 1998). Figure 39 shows that the application of deicer salts has increased since 1940 from 200,000 tons to 10 to 20 million tons per year in recent years (Salt Institute, 2001). Several U.S. and Canadian studies indicate severe inputs of road salts on water quality and aquatic life (Environment Canada, 2001 and Novotny *et al.*, 1999).

Table 43: Use and Water Quality Effect of Snowmelt Deicers (Ohrel, 1995; Sills and Blakeslee, 1992)

Deicer	Description	Use	Water Quality Effect
Chlorides	Chloride based deicer usually combined with Na, Ca or Mg	Road Deicer and Residential Use	Cl complexes can release heavy metals, affect soil permeability, impacts to drinking water, potential toxic effects to small streams
Urea	Nitrogen-based fertilizer product	Used as alternative to glycol	Increased nitrogen in water and potential toxicity to organisms
Ethylene Glycol	Petroleum based organic compounds, similar to antifreeze	Used at airports for deicing planes	Toxicity effects, high BOD and COD, hazardous air pollutant

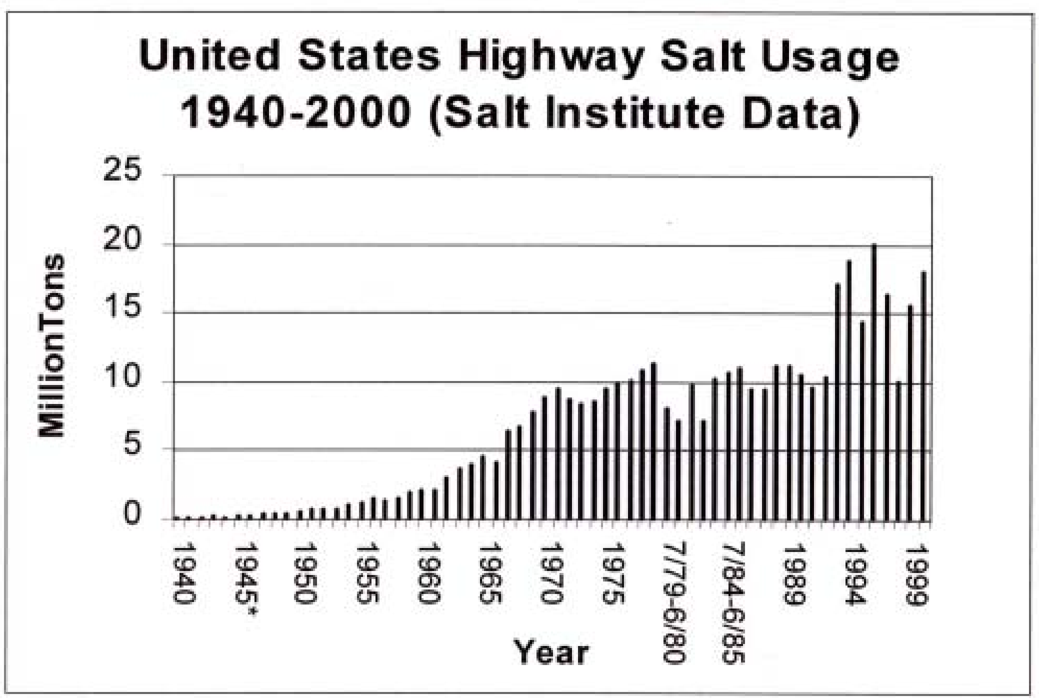


Figure 39: U.S. Highway Salt Usage Data (Salt Institute, 2001)

4.12.1 Concentrations

Chloride concentrations in snowmelt runoff depend on the amount applied and the dilution in the receiving waters. Data for snowmelt and stormwater runoff from several studies are presented in Table 44. For example, chloride concentrations in Lincoln Creek in Wisconsin were 1,612 mg/l in winter snowmelt runoff, as compared to 40 mg/l in non-winter runoff (Novotny *et al.*, 1999 and Masterson and Bannerman, 1994). Chloride concentrations in the range of 2,000 to 5,000 mg/l have been reported for Canadian streams (Environment Canada, 2001). Novotny *et al.* (1999) monitored chloride concentrations in snowmelt near Syracuse, New York and found that residential watersheds had higher chloride concentrations than rural watersheds.

Concentrations of glycol in stormwater runoff are also highly variable and depend on the amount of deicer used, the presence of a recovery system, and the nature of the precipitation event. Corsi *et al*. (2001) monitored streams receiving stormwater runoff from a Wisconsin airport. They found concentrations of propylene glycol as high as 39,000 mg/l at airport outfall sites during deicing operations and concentrations of up to 960 mg/l during low-flow sampling at an airport outfall site.

4.12.2 Impacts of Deicers on Streams

Chloride levels can harm aquatic and terrestrial life and contaminate groundwater and drinking water supplies (Ohrel, 1995). Generally, chloride becomes toxic to many organisms when it reaches concentrations of 500 to1,000 mg/l (Environment Canada, 2001). These concentrations are common in small streams in snow regions, at least for short periods of time. Many plant species are relatively intolerant to high salt levels in wetland swales and roadside corridors. Fish are also negatively affected by high chloride concentrations, with sensitivity as low as 600 mg/l for some species (Scott and Wylie, 1980).

Table 45 compares the maximum chloride concentrations for various water uses in eight states (USEPA, 1988). Snowmelt chloride concentrations typically exceed these levels.

Table 44: EMCs for Chloride in Snowmelt and Stormwater Runoff in Urban Areas

Form of Runoff	EMCs (mg/l) Mean	Number of Events	Sources	Location
Snowmelt	116*	49	Oberts, 1994	MN
	2119	N/R	Sherman, 1998	Ontario
	1267 **R** 474 **U**	N/R	Novotny *et al.*, 1999	NY
	1612	N/R	Masterson and Bannerman, 1994	WI
	397	282	Environment Canada, 2001	Ontario, Canada
Non-winter Storm Event	42	61	Brush *et al.*, 1995	TX
	45	N/R	Sherman, 1998	Ontario
	40.5	N/R	Masterson and Bannerman, 1994	WI

*N/R = Not Reported, R = residential, U = urban, * = Median*

Chloride is a concern in surface drinking water systems because it can interfere with some of the treatment processes and can cause taste problems at concentrations as low as 250 mg/l. Chloride is also extremely difficult to remove once it enters the water.

Glycol-based deicers have been shown to be highly toxic at relatively low concentrations in streams receiving airport runoff. These deicers contain many proprietary agents, which may increase their toxicity and also make it very difficult to set standards for their use (Hartwell *et al.*, 1995). Corsi *et al.* (2001) observed acute toxicity of *Ceriodaphnia dubia*, *Pimephelas promelax*, *Hyalela azteca*, and *Chironimus tentans* in Wisconsin streams that experienced propylene glycol concentrations of 5,000 mg/l or more. Chronic toxicity was observed for *Ceriodaphnia dubia* and *Pimephelas promelax* at propylene glycol concentrations of 1,500 mg/l in the same study. In addition, glycol exerts an extremely high BOD on receiving waters, which can quickly reduce or eliminate dissolved oxygen. Glycol can also be toxic to small animals that are attracted by its sweet taste (Novotny *et al.*, 1999).

As with many urban pollutants, the effects of chloride can be diluted in larger waterbodies. In general, small streams are more likely to experience chloride effects, compared to rivers, which have a greater dilution ability.

4.12.3 Sources and Source Areas of Deicers

The main sources for deicers in urban watersheds include highway maintenance crews, airport deicing operations, and homeowner applications. Direct road application is the largest source of chloride, by far. Source areas include roads, parking lots, sidewalks, storm drains, airport runways, and snow collection areas. Because deicers are applied to paved surfaces, the primary means of transport to streams is through stormwater and meltwater runoff. Therefore, concentrations of deicer compounds are typically associated with factors such as road density or traffic patterns.

Table 45: Summary of State Standards for Salinity of Receiving Waters (USEPA, 1988)

State	Limiting Concentration (mg/l)	Beneficial Use
CO	250*	Drinking water
IL	500	General water supply
	250	Drinking water
IN	500	Drinking water
MA	250	Class A waters
MN	250	Drinking water
	500	Class A fishing and recreation
OH	250	Drinking water
SD	250	Drinking water
	100	Fish propagation
VA	250	Drinking water

** Monthly average*

4.13 Conclusion

IC collects and accumulates pollutants deposited from the atmosphere, leaked from vehicles, or derived from other sources. The pollutants build up over time but are washed off quickly during storms and are often efficiently delivered to downstream waters. This can create water quality problems for downstream rivers, lakes and estuaries.

As a result of local and national monitoring efforts, we now have a much better understanding of the nature and impacts of stormwater pollution. The typical sample of urban stormwater is characterized by high levels of many common pollutants such as sediment, nutrients, metals, organic carbon, hydrocarbons, pesticides, and fecal coliform bacteria. Other pollutants that have more recently become a concern in urban areas include MTBE, deicers, and the pathogens *Cryptosporidium* and *Giardia*. Concentrations of most stormwater pollutants can be characterized, over the long run, by event mean storm concentrations. Monitoring techniques have also allowed researchers to identify source areas for pollutants in the urban environment, including stormwater hotspots, which generate higher pollutant loads than normal development.

In general, most monitoring data shows that mean pollutant storm concentrations are higher in urban watersheds than in non-urban ones. For many urban pollutants, EMCs can be used to predict stormwater pollutant loads for urban watersheds, using IC as the key predictive variable. While a direct relationship between IC and pollutant concentrations does not usually exist, IC directly influences the volume of stormwater and hence, the total load. A few exceptions are worth noting. MTBE, deicers, and PAH appear to be related more to traffic or road density than IC. Additionally, MTBE and PAH concentrations may be greater at hotspot source areas, which are not always widely or uniformly distributed across a watershed. Pesticides, bacteria and pathogens are often associated with turf areas rather than IC. Bacteria and pathogen sources also include direct inputs from wildlife and inappropriate sewage discharges that are not uniformly distributed across a watershed and are not directly related to IC.

Further research into the relationship between stormwater pollutant loads and other watershed indicators may be helpful. For example, it would be interesting to see if turf cover is a good indicator of stream quality for impacted streams. Other important watershed indicators worth studying are the influence of watershed treatment practices, such as stormwater practices and stream buffers.

The direct effects of stormwater pollutants on aquatic systems appears to be a function of the size of the receiving water and the initial health of the aquatic community. For example, a small urban stream receiving high stormwater pollutant concentrations would be more likely to experience impacts than a large river, which is diluted by other land uses. Likewise, organisms in sensitive streams should be more susceptible to stormwater pollutants than pollution-tolerant organisms found in non-supporting streams.

Overall, the following conclusions can be made:

- Sediment, nutrient and trace metal loads in stormwater runoff can be predicted as a function of IC, although concentrations are not tightly correlated with watershed IC.
- Violations of bacteria standards are indirectly associated with watershed IC.
- It is not clear whether loads of hydrocarbons, pesticides or chlorides can be predicted on the basis of IC at the small watershed level.
- More research needs to be conducted to evaluate the usefulness of other watershed indicators to predict stormwater pollutant loads. For example, traffic, road density or hotspots may be useful in predicting MTBE, deicer and hydrocarbon loads. Also, watershed turf cover may be useful in predicting pesticide and bacterial loads.

- Most research on pollutants in stormwater runoff has been conducted at the small watershed level. Additional research is needed to evaluate the impact of watershed treatment, such as stormwater and buffer practices to determine the degree to which these may change stormwater concentrations or loads.
- Regional differences are evident for many stormwater pollutants, and these appear to be caused by either differences in rainfall frequency or snowmelt.

Chapter 5: Biological Impacts of Impervious Cover

This chapter reviews research on the impact of urbanization on the aquatic community, focusing on aquatic insects, fish, amphibians, freshwater mussels, and freshwater wetlands. Specifically, the relationship between the health of the aquatic community and the amount of watershed IC is analyzed within the context of the Impervious Cover Model (ICM).

The chapter is organized as follows:

5.1 Introduction
5.2 Indicators and General Trends
5.3 Effects on Aquatic Insect[1] Diversity
5.4 Effects on Fish Diversity
5.5 Effects on Amphibian Diversity
5.6 Effects on Wetland Diversity
5.7 Effects on Freshwater Mussel Diversity
5.8 Conclusion

5.1 Introduction

A number of studies, crossing different ecoregions and utilizing various techniques, have examined the link between watershed urbanization and its impact on stream and wetland biodiversity. These studies reveal that a relatively small amount of urbanization has a negative effect on aquatic diversity, and that as watersheds become highly urban, aquatic diversity becomes extremely degraded. As documented in prior chapters, hydrologic, physical, and water quality changes caused by watershed urbanization all stress the aquatic community and collectively diminish the quality and quantity of available habitat. As a result, these stressors generally cause a decline in biological diversity, a change in trophic structure, and a shift towards more pollution-tolerant organisms.

Many different habitat conditions are critical for supporting diverse aquatic ecosystems. For example, streambed substrates are vulnerable to deposition of fine sediments, which affects spawning, egg incubation and fry-rearing. Many aquatic insect species shelter in the large pore spaces among cobbles and boulders, particularly within riffles. When fine sediment fills these pore spaces, it reduces the quality and quantity of available habitat. The aquatic insect community is typically the base of the food chain in streams, helps break down organic matter and serves as a food source for juvenile fish.

Large woody debris (LWD) plays a critical role in the habitat of many aquatic insects and fish. For example, Bisson *et al.* (1988) contend that no other structural component is more important to salmon habitat than LWD, especially in the case of juvenile coho salmon. Loss of LWD due to the removal of stream side vegetation can significantly hinder the survival of more sensitive aquatic species. Since LWD creates different habitat types, its quality and quantity have been linked to salmonid rearing habitat and the ability of multiple fish species to coexist in streams.

The number of stream crossings (e.g., roads, sewers and pipelines) has been reported to increase directly in proportion to IC (May *et al*., 1997). Such crossings can become partial or total barriers to upstream fish migration, particularly if the stream bed downcuts below the fixed elevation of a culvert or pipeline. Fish barriers can prevent migration and recolonization of aquatic life in many urban streams.

Urbanization can also increase pollutant levels and stream temperatures. In particular, trace metals and pesticides often bind to sediment particles and may enter the food chain, particularly by aquatic insects that collect and filter particles. While in-stream data is rare, some data are available for ponds. A study of trace

[1]*Throughout this chapter, the term "aquatic insects" is used rather than the more cumbersome but technically correct "benthic macroinvertebrates."*

metal bioaccumulation of three fish species found in central Florida stormwater ponds discovered that trace metal levels were significantly higher in urban ponds than in non-urban control ponds, often by a factor of five to 10 (Campbell, 1995; see also Karouna-Renier, 1995). Although typical stormwater pollutants are rarely acutely toxic to fish, the cumulative effects of sublethal pollutant exposure may influence the stream community (Chapter 4).

Table 46 summarizes some of the numerous changes to streams caused by urbanization that have the potential to alter aquatic biodiversity. For a comprehensive review of the impacts of urbanization on stream habitat and biodiversity, the reader should consult Wood and Armitage (1997) and Hart and Finelli (1999).

Table 46: Review of Stressors to Urban Streams and Effects on Aquatic Life

Stream Change	Effects on Organisms
Increased flow volumes/ Channel forming storms	Alterations in habitat complexity Changes in availability of food organisms, related to timing of emergence and recovery after disturbance Reduced prey diversity Scour-related mortality Long-term depletion of LWD Accelerated streambank erosion
Decreased base flows	Crowding and increased competition for foraging sites Increased vulnerability to predation Increased fine sediment deposition
Increase in sediment transport	Reduced survival of eggs and alevins, loss of habitat due to deposition Siltation of pool areas, reduced macroinvertebrate reproduction
Loss of pools and riffles	Shift in the balance of species due to habitat change Loss of deep water cover and feeding areas
Changes in substrate composition	Reduced survival of eggs Loss of inter-gravel fry refugial spaces Reduced aquatic insect production
Loss of LWD	Loss of cover from predators and high flows Reduced sediment and organic matter storage Reduced pool formation and organic substrate for aquatic insects
Increase in temperature	Changes in migration patterns Increased metabolic activity, increased disease and parasite susceptibility Increased mortality of sensitive fish
Creation of fish blockages	Loss of spawning habitat for adults Inability to reach overwintering sites Loss of summer rearing habitat, Increased vulnerability to predation
Loss of vegetative rooting systems	Decreased channel stability Loss of undercut banks Reduced streambank integrity
Channel straightening or hardening	Increased stream scour Loss of habitat complexity
Reduction in water quality	Reduced survival of eggs and alevins Acute and chronic toxicity to juveniles and adult fish Increased physiological stress
Increase in turbidity	Reduced survival of eggs Reduced plant productivity Physiological stress on aquatic organisms
Algae blooms	Oxygen depletion due to algal blooms, increased eutrophication rate of standing waters

5.2 Indicators and General Trends

Stream indicators are used to gauge aquatic health in particular watersheds. The two main categories of stream indicators are **biotic** and **development** indices. **Biotic** indices use stream diversity as the benchmark for aquatic health and use measures, such as species abundance, taxa richness, EPT Index, native species, presence of pollution-tolerant species, dominance, functional feeding group comparisons, or proportion with disease or anomalies. **Development** indices evaluate the relationship between the degree of watershed urbanization and scores for the biotic indices. Common development indices include watershed IC, housing density, population density, and percent urban land use.

5.2.1 Biological Indicators

Biotic indices are frequently used to measure the health of the aquatic insect or fish community in urban streams. Because many aquatic insects have limited migration patterns or a sessile mode of life, they are particularly well-suited to assess stream impacts over time. Aquatic insects integrate the effects of short-term environmental variations, as most species have a complex but short life cycle of a year or less. Sensitive life stages respond quickly to environmental stressors, but the overall community responds more slowly. Aquatic insect communities are comprised of a broad range of species, trophic levels and pollution tolerances, thus providing strong information for interpreting cumulative effects. Unlike fish, aquatic insects are abundant in most small, first and second order streams. Individuals are relatively easy to identify to family level, and many “intolerant” taxa can be identified to lower taxonomic levels with ease.

Fish are good stream indicators over longer time periods and broad habitat conditions because they are relatively long-lived and mobile. Fish communities generally include a range of species that represents a variety of trophic levels (omnivores, herbivores, insectivores, planktivores, and piscivores). Fish tend to integrate the effects of lower trophic levels; thus, their community structure reflects the prevailing food sources and habitat conditions. Fish are relatively easy to collect and identify to the species level. Most specimens can be sorted and identified in the field by experienced fisheries scientists and subsequently released unharmed.

A review of the literature indicates that a wide variety of metrics are used to measure the aquatic insect and fish community. Community indices, such as the Index of Biotic Integrity (IBI) for fish and the Benthic Index of Biotic Integrity (B-IBI) for the aquatic insect community are a weighted combination of various metrics that typically characterize the community from “excellent” to “poor.” Common metrics of aquatic community are often based on a composite of measures, such as species richness, abundance, tolerance, trophic status, and native status. Combined indices (C-IBI) measure both fish and aquatic insect metrics and a variety of physical habitat conditions to classify streams. Table 47 lists several common metrics used in stream assessments. It should be clearly noted that community and combined indices rely on different measurements and cannot be directly compared. For a comprehensive review of aquatic community indicators, see Barbour *et al.*(1999).

5.2.2 Watershed Development Indices

Watershed IC, housing density, population density, and percent urban land have all been used as indices of the degree of watershed development. In addition, reverse indicators such as percent forest cover and riparian continuity have also been used. The majority of studies so far have used IC to explore the relationship between urbanization and aquatic diversity. Percent urban land has been the second most frequently used indicator to describe the impact of watershed development. Table 48 compares the four watershed development indices and the thresholds where significant impacts to aquatic life are typically observed.

Table 47: Examples of Biodiversity Metrics Used to Assess Aquatic Communities		
Measurement	**Applied to:**	**Definition of Measurement**
Abundance	Fish, Aquatic Insects	Total number of individuals in a sample; sometimes modified to exclude tolerant species.
Taxa Richness	Fish, Aquatic Insects	Total number of unique taxa identified in a sample. Typically, an increase in taxa diversity indicates better water and habitat quality.
EPT Index	Aquatic Insects	Taxa belonging to the following three groups: *Ephemeroptera* (mayflies), *Plecoptera* (stoneflies), *Trichoptera* (caddisflies). Typically, species in these orders are considered to be pollution-intolerant taxa and are generally the first to disappear with stream quality degradation.
Native Status	Fish	Native vs. non-native taxa in the community.
Specific Habitat	Fish	Riffle benthic insectivorous individuals. Total number of benthic insectivores. Often these types of individuals, such as darters, sculpins, and dace are found in high velocity riffles and runs and are sensitive to physical habitat degradation.
		Minnow species Total number of minnow species present. Often used as an indicator of pool habitat quality. Includes all species present in the family Cyprinidae, such as daces, minnows, shiners, stonerollers, and chubs.
Tolerant Species	Fish, Aquatic Insects	The total number of species sensitive to and the number tolerant of degraded conditions. Typically, intolerant species decline with decreasing water quality and stream habitat. A common high pollution-tolerant species that is frequently used is Chironomids.
Dominance	Fish, Aquatic Insects	The proportion of individuals at each station from the single most abundant taxa at that particular station. Typically, a community dominated by a single taxa may be indicative of stream degradation.
Functional Feeding Group Comparisons	Fish	Omnivores/ Generalists: The proportion of individuals characterized as omnivores or generalists to the total number of individuals. Typically, there is a shift away from specialized feeding towards more opportunistic feeders under degraded conditions as food sources become unreliable.
		Insectivores: The proportion of individuals characterized as insectivores to the total number of individuals. Typically, the abundance of insectivores decreases relative to increasing stream degradation.
	Aquatic Insects	Others: The proportion of individuals characterized as shredders, scrapers, or filter feeders to the total number of individuals. Typically, changes in the proportion of functional feeders characterized as shredders can be reflective of contaminated leaf matter. In addition, an overabundance of scrapers over filterers can be indicative of increased benthic algae.
Disease/ Anomalies	Fish	Proportion of individuals with signs of disease or abnormalities. This is ascertained through gross external examination for abnormalities during the field identification process. Typically, this metric assumes that incidence of disease and deformities increases with increasing stream degradation.
** This table is not meant to provide a comprehensive listing of metrics used for diversity indices; it is intended to provide examples of types of measures used in biological stream assessments (see Barbour* et al., *1999).*		

5.2.3 General Trends

Most research suggests that a decline in both species abundance and diversity begins at or around 10% watershed IC (Schueler, 1994a). However, considerable variations in aquatic diversity are frequently observed from five to 20% IC, due to historical alterations, the effectiveness of watershed management, prevailing riparian conditions, co-occurrence of stressors, and natural biological variation (see Chapter 1).

Figures 40 through 42 display the negative relationship commonly seen between biotic indices and various measures of watershed development. For example, stream research in the Maryland Piedmont indicated that IC was the best predictor of stream condition, based on a combined fish and aquatic insect IBI (MNCPPC, 2000). In general, streams with less than 6% watershed IC were in "excellent" condition, whereas streams in "good" condition had less than 12% IC, and streams in "fair" condition had less than 20%. Figure 40 shows the general boundaries and typical variation seen in MNCPPC stream research.

Figure 41 illustrates that B-IBI scores and Coho Salmon/Cutthroat Trout Ratio are a function of IC for 31 streams in Puget Sound, Washington. The interesting finding was that "good" to "excellent" B-IBI scores (greater than 25) were reported in watersheds that had less than 10% IC, with eight notable outliers. These outliers had greater IC (25 to 35%) but similar B-IBI scores. These outliers are unique in that they had a large upstream wetland and/or a large, intact riparian corridor upstream (i.e. >70% of stream corridor had buffer width >100 feet).

Figure 42 depicts the same negative relationship between watershed urbanization and fish-IBI scores but uses population density as the primary metric of development (Dreher, 1997). The six-county study area included the Chicago metro area and outlying rural watersheds. Significant declines in fish-IBI scores were noted when population density exceeded 1.5 persons per acre.

The actual level of watershed development at which an individual aquatic species begins to decline depends on several variables, but may be lower than that indicated by the ICM. Some researchers have detected impacts for individual aquatic species at watershed IC levels as low as 5%. Other research has suggested that the presence of certain stressors, such as sewage treatment plant discharges (Yoder and Miltner, 2000) or construction sites (Reice, 2000) may alter the ICM and lower the level of IC at which biodiversity impacts become evident.

Table 48: Alternate Land Use Indicators and Significant Impact Levels (Brown, 2000; Konrad and Booth, 2002)			
Land Use Indicator	**Level at which Significant Impact Observed**	**Typical Value for Low Density Residential Use**	**Comments**
% IC	10-20%	10%	Most accurate; highest level of effort and cost
Housing Density	>1 unit/acre	1 unit/acre	Low accuracy in areas of substantial commercial or industrial development; less accurate at small scales
Population Density	1.5 to 8+ people/acre	2.5 people/acre	Low accuracy in areas of substantial commercial or industrial development; less accurate at small scales
% Urban Land Use	33% (variable)	10-100%	Does not measure intensity of development; moderately accurate at larger watershed scales
Road Density	5 miles/square mile	2 miles/square mile	Appears to be a potentially useful indicator

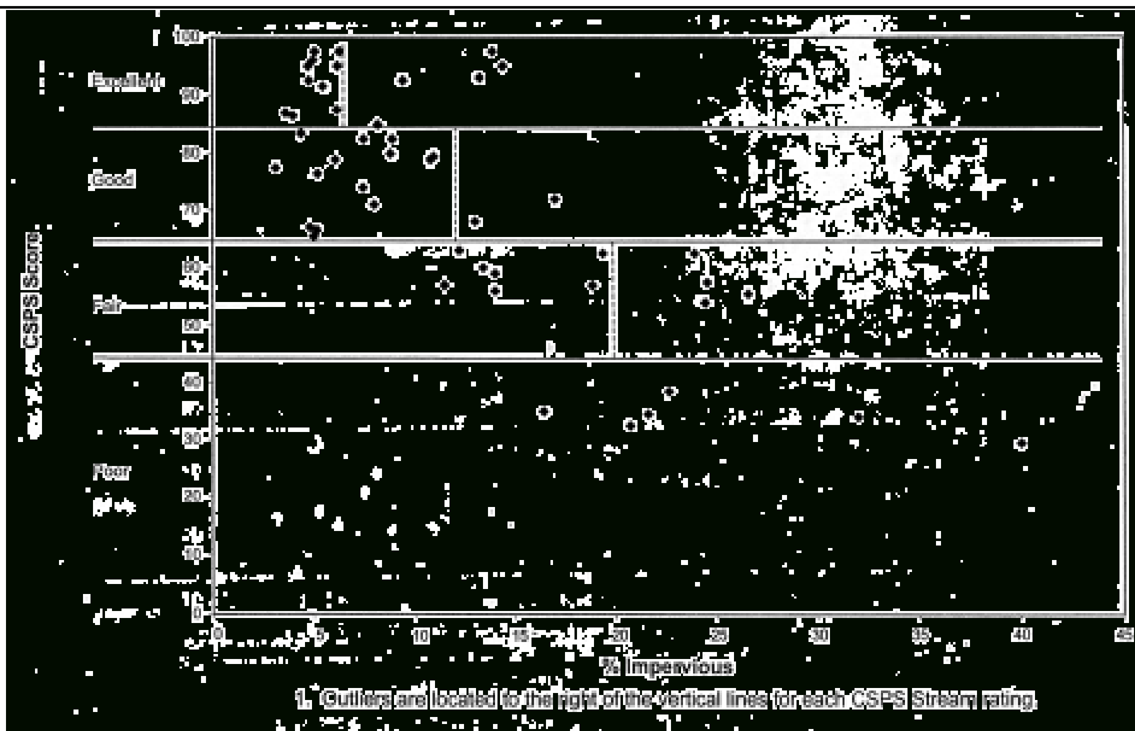


Figure 40: Combined Fish and Benthic IBI vs. IC in Maryland Piedmont Streams (MNCPPC, 2000)

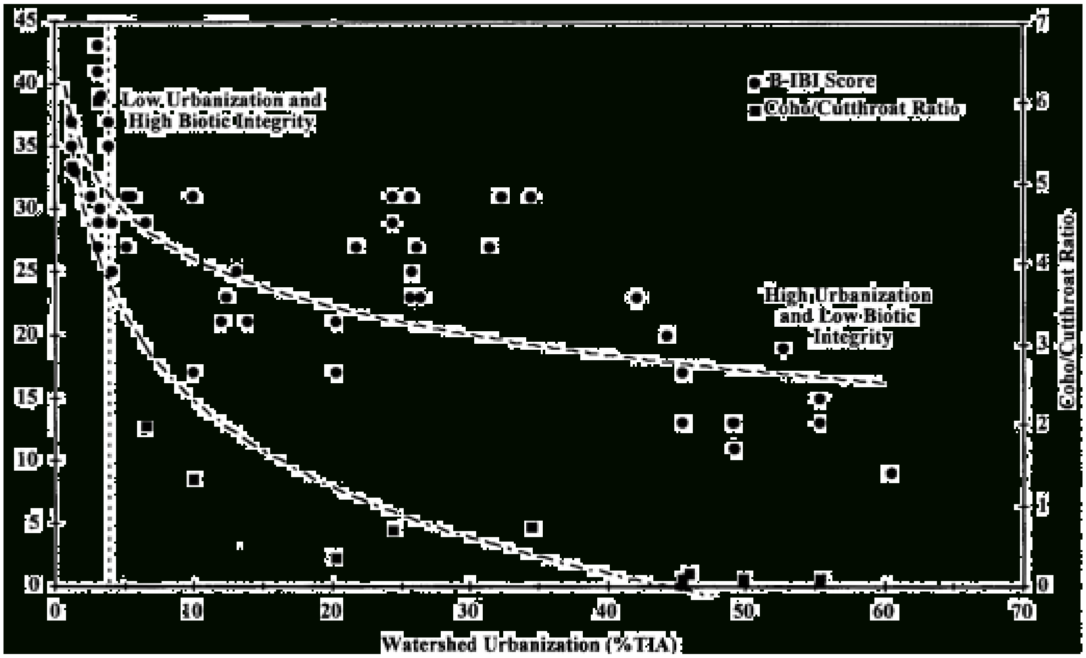


Figure 41: Relationship Between B-IBI, Coho/Cutthroat Ratios, and Watershed IC in Puget Sound Streams (Horner *et al.*, 1997)

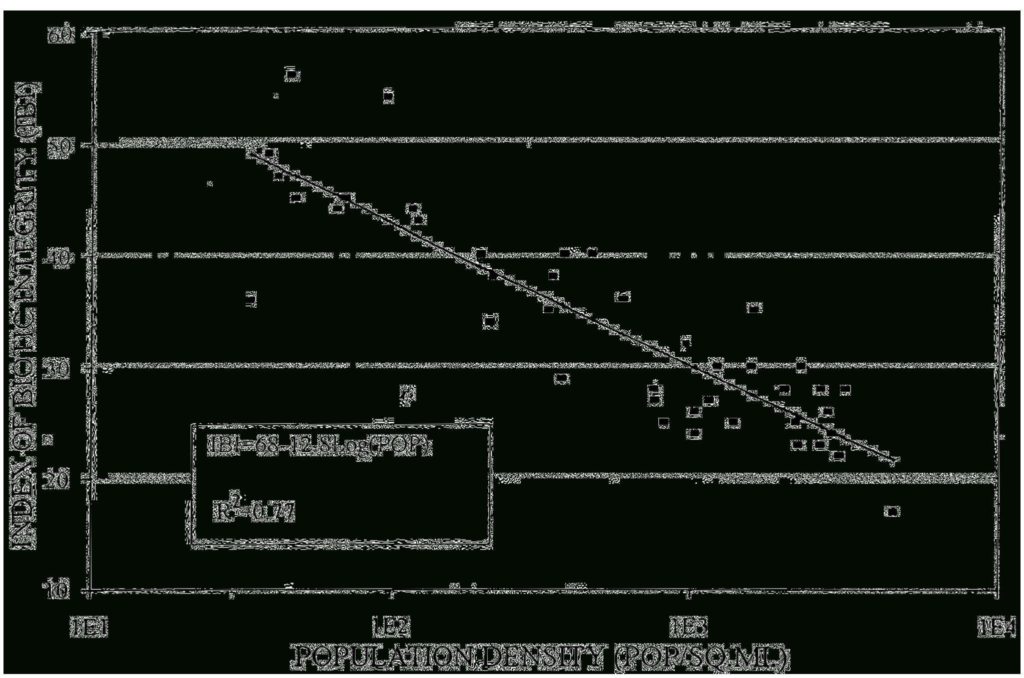


Figure 42: Index for Biological Integrity as a Function of Population Density in Illinois (Dreher, 1997)

5.3 Effects on Aquatic Insect Diversity

The diversity, richness and abundance of the aquatic insect community is frequently used to indicate urban stream quality. Aquatic insects are a useful indicator because they form the base of the stream food chain in most regions of the country. For this reason, declines or changes in aquatic insect diversity are often an early signal of biological impact due to watershed development. The aquatic insect community typically responds to increasing development by losing species diversity and richness and shifting to more pollution-tolerant species. More than 30 studies illustrate how IC and urbanization affect the aquatic insect community. These are summarized in Tables 49 and 50.

5.3.1 Findings Based on IC Indicators

Klein (1979) was one of the first researchers to note that aquatic insect diversity drops sharply in streams where watershed IC exceeded 10 to 15%. While "good" to "fair" diversity was noted in all headwater streams with less than 10% IC, nearly all streams with 12% or more watershed IC recorded "poor" diversity. Other studies have confirmed this general relationship between IC and the decline of aquatic insect species diversity. Their relationships have been an integral part in the development of the ICM. The sharp drop in aquatic insect diversity at or around 12 to 15% IC was also observed in streams in the coastal plain and Piedmont of Delaware (Maxted and Shaver, 1997).

Impacts at development thresholds lower than 10% IC have also been observed by Booth (2000), Davis (2001), Horner *et al.* (1997) and Morse (2001). There seems to be a general recognition that the high levels of variability observed below 10% IC indicate that other factors, such as riparian condition, effluent discharges, and pollution legacy may be better indicators of aquatic insect diversity (Horner and May, 1999; Kennen, 1999; Steedman, 1988; Yoder *et al.*, 1999).

The exact point at which aquatic insect diversity shifts from fair to poor is not known with absolute precision, but it is clear that few, if any, urban streams can support diverse aquatic insect communities with more than 25% IC. Indeed, several researchers failed to find aquatic insect communities with good or excellent diversity in any highly urban stream (Table 52). Indeed, MNCPPC (2000) reported that all streams with more than 20% watershed IC were rated as "poor."

Several good examples of the relationship between IC and B-IBI scores are shown in Figures 43 through 45. Figure 43 depicts the general trend line in aquatic insect diversity as IC increased at 138 stream sites in Northern Virginia (Fairfax County, 2001). The survey study concluded that stream degradation occurred at low levels of IC, and that older developments lacking more efficient site design and stormwater controls tended to have particularly degraded streams. Figures 44 and 45 show similar trends in the relationship between IC and aquatic insect B-IBI scores in Maryland and Washington streams. In particular, note the variability in B-IBI scores observed below 10% IC in both research studies.

Often, shift in the aquatic insect community from pollution-sensitive species to pollution-tolerant species occurs at relatively low IC levels (<10%). This shift is often tracked using the EPT metric, which evaluates sensitive species found in the urban stream community in the orders of *Ephemeroptera* (mayflies), *Plecoptera* (stoneflies), and *Trichoptera* (caddisflies). EPT species frequently disappear in urban streams and are replaced by more pollution-tolerant organisms, such as chironomids, tubificid worms, amphipods and snails.

In undisturbed streams, aquatic insects employ specialized feeding strategies, such as shredding leaf litter, filtering or collecting organic matter that flows by, or preying on other insects. These feeding guilds are greatly reduced in urban streams and are replaced by grazers, collectors and deposit feeders. Maxted and Shaver (1997) found that 90% of sensitive

Table 49: Recent Research Examining the Relationship Between IC and Aquatic Insect Diversity in Streams			
Index	**Key Finding (s)**	**Source**	**Location**
Community Index	Three years stream sampling across the state at 1000 sites found that when IC was >15%, stream health was never rated good based on a C-IBI.	Boward *et al.*, 1999	MD
Community Index	Insect community and habitat scores were all ranked as poor in five subwatersheds that were greater than 30% IC.	Black and Veatch, 1994	MD
Community Index	Puget sound study finds that some degradation of aquatic invertebrate diversity can occur at any level of human disturbance (at least as measured by IC). 65% of watershed forest cover usually indicates a healthy aquatic insect community.	Booth, 2000	WA
Community Index	In a Puget Sound study, the steepest decline of B-IBI was observed after 6% IC. There was a steady decline, with approximately 50% reduction in B-IBI at 45% IC.	Horner *et al.*, 1997	WA
Community Index	B-IBI decreases with increasing urbanization in study involving 209 sites, with a sharp decline at 10% IC. Riparian condition helps mitigate effects.	Steedman, 1988	Ontario
Community Index	Wetlands, forest cover and riparian integrity act to mitigate the impact of IC on aquatic insect communities.	Horner *et al.*, 2001	WA, MD, TX
Community Index	B-IBI declines for aquatic insect with increasing IC at more than 200 streams.	Fairfax Co., 2001	VA
Community Index	Two-year stream study of eight Piedmont watersheds reported B-IBI scores declined sharply at an IC threshold of 15-30%.	Meyer and Couch,2000	GA
Community Index	Montgomery County study; subwatersheds with <12% IC generally had streams in good to excellent condition based on a combined fish and aquatic insect IBI. Watersheds with >20% IC had streams in poor condition.	MNCPPC, 2000	MD
Community Index	Study of 1st, 2nd, and 3rd order streams in the Patapsco River Basin showed negative relationship between B-IBI and IC.	Dail *et al.*, 1998	MD
Community Index	While no specific threshold was observed, impacts were seen at even low levels of IC. B-IBI values declined with increasing IC, with high scores observed only in reaches with <5% IC or intact riparian zones or upstream wetlands.	Horner and May, 1999	WA
Community Index	The C-IBI also decreased by 50% at 10-15% IC. These trends were particularly strong at low-density urban sites (0-30% IC).	Maxted and Shaver, 1997	DE
Diversity	In both coastal plain and Piedmont streams, a sharp decline in aquatic insect diversity was found around 10-15% IC.	Shaver *et al.*, 1995	DE
Diversity	In a comparison of Anacostia subwatersheds, there was significant decline in the diversity of aquatic insects at 10% IC.	MWCOG, 1992	DC
Diversity	In several dozen Piedmont headwater streams, aquatic diversity declined significantly beyond 10-12% IC.	Klein, 1979	MD
EPT Value	In a 10 stream study with watershed IC ranging from three to 30%, a significant decline in EPT values was reported as IC increased ($r2 = 0.76$).	Davis, 2001	MO
Sensitive Species	In a study of 38 wadeable, non-tidal streams in the urban Piedmont, 90% of sensitive organisms were eliminated from the benthic community after watershed IC reaches 10-15%.	Maxted and Shaver, 1997	DE
Species Abundance EPT values	For streams draining 20 catchments across the state, an abrupt decline in species abundance and EPT taxa was observed at approximately 6% IC.	Morse, 2001	ME

Table 50: Recent Research Examining the Relationship of Other Indices of Watershed Development on Aquatic Insect Diversity in Streams

Biotic	Key Finding (s)	Source	Location
Percent Urban Land use			
Community Index	Study of 700 streams in 5 major drainage basins found that the amount of urban land and total flow of municipal effluent were the most significant factors in predicting severe impairment of the aquatic insect community. Amount of forested land in drainage area was inversely related to impairment severity.	Kennen, 1999	NJ
Community Index	All 40 urban sites sampled had fair to very poor B-IBI scores, compared to undeveloped reference sites.	Yoder, 1991	OH
Community Index	A negative correlation between B-IBI and urban land use was noted. Community characteristics show similar patterns between agricultural and forested areas the most severe degradation being in urban and suburban areas.	Meyer and Couch, 2000	GA
EPT Value, Diversity, Community Index	A comparison of three stream types found urban streams had lowest diversity and richness. Urban streams had substantially lower EPT scores (22% vs 5% as number of all taxa, 65% vs 10% as percent abundance) and IBI scores in the poor range.	Crawford and Lenat, 1989	NC
Sensitive Species	Urbanization associated with decline in sensitive taxa, such as mayflies, caddisflies and amphipods while showing increases in oligochaetes.	Pitt and Bozeman, 1982	CA
Sensitive Species	Dramatic changes in aquatic insect community were observed in most urbanizing stream sections. Changes include an abundance of pollution-tolerant aquatic insect species in urban streams.	Kemp and Spotila, 1997	PA
Diversity	As watershed development levels increased, the aquatic insect diversity declined.	Richards *et al.*, 1993	MN
Diversity	Significant negative relationship between number of aquatic insect species and degree of urbanization in 21 Atlanta streams.	Benke *et al.*, 1981	GA
Diversity	Drop in insect taxa from 13 to 4 was noted in urban streams.	Garie and McIntosh, 1986	NJ
Diversity	Aquatic insect taxa were found to be more abundant in non-urban reaches than in urban reaches of the watershed.	Pitt and Bozeman, 1982	CA
Diversity	A study of five urban streams found that as watershed land use shifted from rural to urban, aquatic insect diversity decreased.	Masterson and Bannerman, 1994	WI
Other Land Use Indicators			
Community Index	Most degraded streams were found in developed areas, particularly older developments lacking newer and more efficient stormwater controls.	Fairfax Co., 2001	VA
Diversity	Urban streams had sharply lower aquatic insect diversity with human population above four persons/acre in northern VA.	Jones and Clark, 1987	VA
EPT Value	Monitoring of four construction sites in three varying regulatory settings found that EPT richness was related to enforcement of erosion and sediment controls. The pattern demonstrated that EPT richness was negatively affected as one moved from upstream to at the site, except for one site.	Reice, 2000	NC
Sensitive Species	In a Seattle study, aquatic insect community shifted to chironomid, oligochaetes and amphipod species that are pollution-tolerant and have simple feeding guild.	Pedersen and Perkins, 1986	WA

species (based on EPT richness, % EPT abundance, and Hilsenhoff Biotic Index) were eliminated from the aquatic insect community when IC exceeded 10 to 15% in contributing watersheds of Delaware streams (Figure 46). In a recent study of 30 Maine watersheds, Morse (2001) found that reference streams with less than 5% watershed IC had significantly more EPT taxa than more urban streams. He also observed no significant differences in EPT Index values among streams with six to 27% watershed IC (Figure 47).

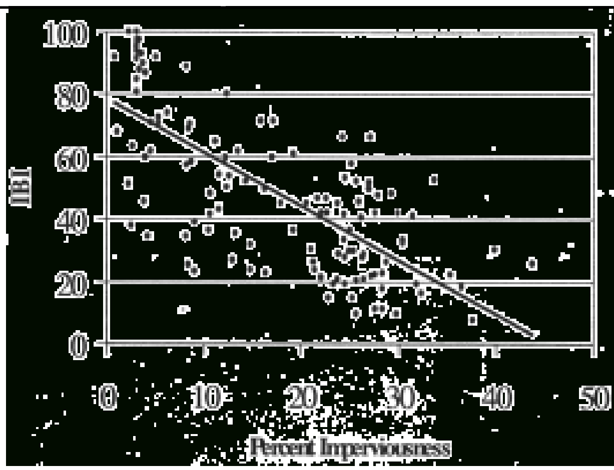


Figure 43: Trend Line Indicating Decline in Benthic IBI as IC Increases in Northern VA Streams (Fairfax County, 2001)

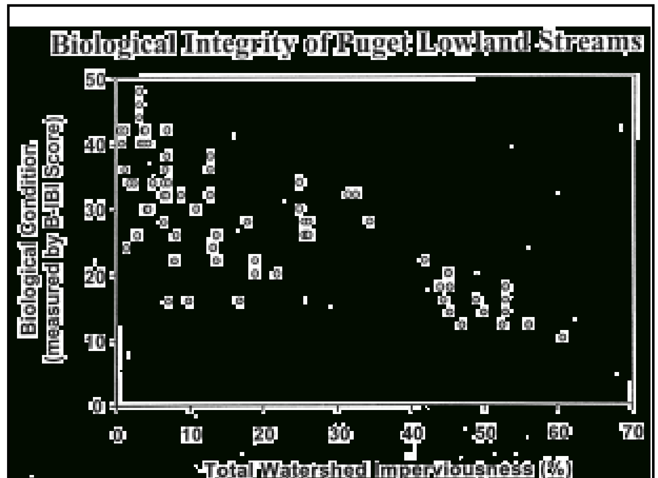


Figure 44: Relationship Between IC and B-IBI Scores in Aquatic Insects in Streams of the Puget Sound Lowlands (Booth, 2000)

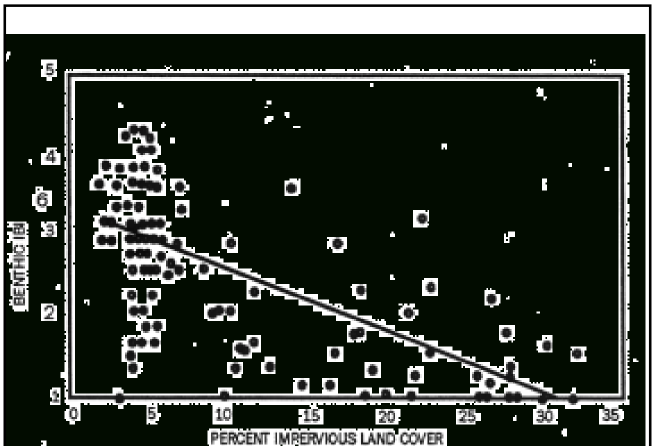


Figure 45: IC and B-IBI at Stream Sites in the Patapsco River Basin, MD (Dail *et al.*, 1998)

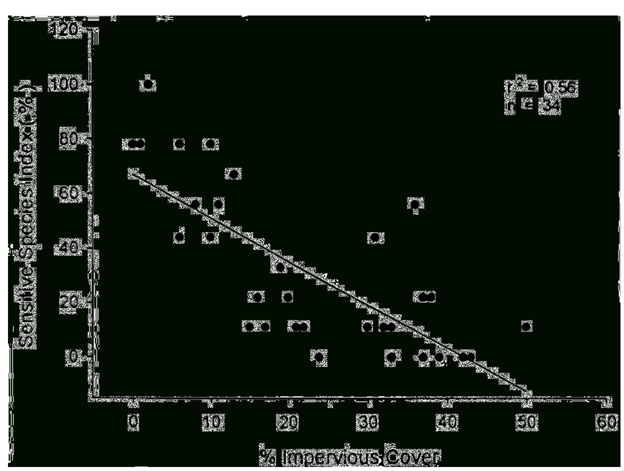


Figure 46: IC vs. Aquatic Insect Sensitivity - EPT Scores in Delaware Streams (Maxted and Shaver, 1997)

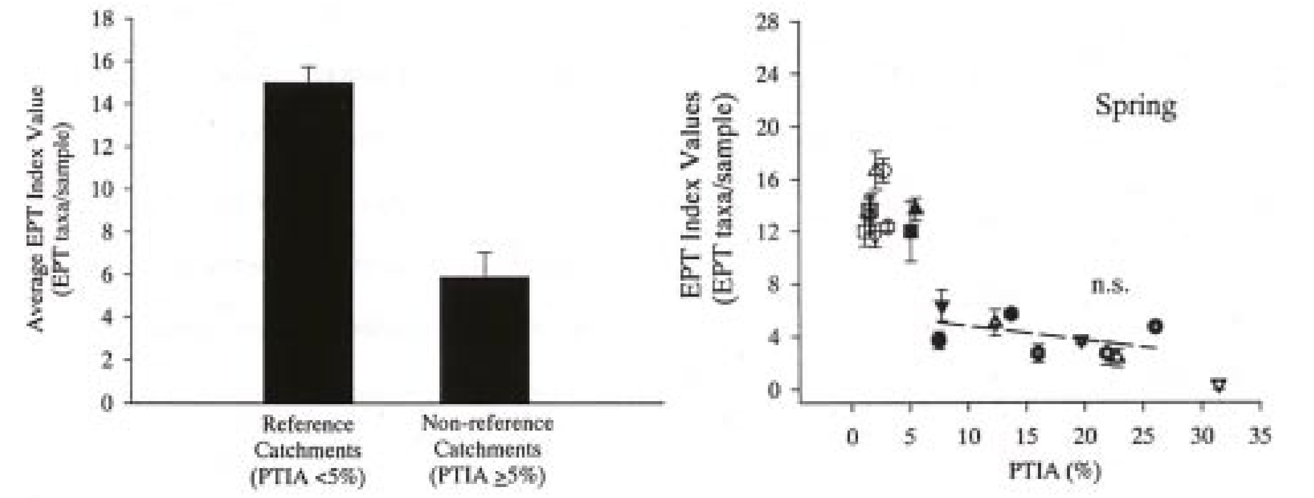


Figure 47: Average and Spring EPT Index Values vs.% IC in 20 Small Watersheds in Maine (Morse, 2001)

5.3.2 Findings Based on Other Development Indicators

Development indices, such as percent urban land use, population density, and forest and riparian cover have also been correlated with changes in aquatic insect communities in urban streams. Declines in benthic IBI scores have frequently been observed in proportion to the percent urban land use in small watersheds (Garie and McIntosh, 1986; Kemp and Spotila, 1997; Kennen, 1999; Masterson and Bannerman, 1994; Richards *et al.*, 1993; USEPA, 1982).

A study in Washington state compared a heavily urbanized stream to a stream with limited watershed development and found that the diversity of the aquatic insect community declined from 13 taxa in reference streams to five taxa in more urbanized streams (Pedersen and Perkins, 1986). The aquatic insect taxa that were lost were poorly suited to handle the variable erosional and depositional conditions found in urban streams. Similarly, a comparison of three North Carolina streams with different watershed land uses concluded the urban watershed had the least taxa and lowest EPT scores and greatest proportion of pollution-tolerant species (Crawford and Lenat, 1989).

Jones and Clark (1987) monitored 22 streams in Northern Virginia and concluded that aquatic insect diversity diminished markedly once watershed population density exceeded four or more people per acre. The population density roughly translates to ½ - 1 acre lot residential use, or about 10 to 20 % IC. Kennen (1999) evaluated 700 New Jersey streams and concluded that the percentage of watershed forest was positively correlated with aquatic insect density. Meyer and Couch (2000) reported a similar cover relationship between aquatic insect diversity and watershed and riparian forest cover for streams in the Atlanta, GA region. A study in the Puget Sound region found that aquatic insect diversity declined in streams once forest cover fell below 65% (Booth, 2000).

5.4 Effects on Fish Diversity

Fish communities are also excellent environmental indicators of stream health. In general, an increase in watershed IC produces the same kind of impact on fish diversity as it does for aquatic insects. The reduction in fish diversity is typified by a reduction in total species, loss of sensitive species, a shift toward more pollution-tolerant species, and decreased survival of eggs and larvae. More than 30 studies have examined the relationship between watershed development and fish diversity; they are summarized in Tables 51 and 52. About half of the research studies used IC as the major index of watershed development, while the remainder used other indices, such as percent urban land use, population density, housing density, and forest cover.

5.4.1 Findings Based on IC Indicators

Recent stream research shows a consistent, negative relationship between watershed development and various measures of fish diversity, such as diversity metrics, species loss and structural changes.

Typically, a notable decline in fish diversity occurs around 10 to 15% watershed IC (Boward *et al.*, 1999; Galli, 1994; Klein, 1979; Limburg and Schmidt, 1990; MNCPPC, 2000; MWCOG, 1992; Steward, 1983). A somewhat higher threshold was observed by Meyer and Couch (2000) for Atlanta streams with 15 to 30% IC; lower thresholds have also been observed (Horner *et al.*, 1997 and May *et al.*, 1997). A typical relationship between watershed IC and fish diversity is portrayed in Figure 48, which shows data from streams in the Patapsco River Basin in Maryland (Dail *et al.*, 1998). Once again, note the variability in fish-IBI scores observed below 10% IC.

Wang *et al.* (1997) evaluated 47 Wisconsin streams and found an apparent threshold around 10% IC. Fish-IBI scores were "good" to "excellent" below this threshold, but were consistently rated as "fair" to "poor." Additionally, Wang documented that the total number of fish species drops sharply when IC increases (Figure 49). Often, researchers also reported that increases in IC were strongly correlated with several fish metrics, such as increases in non-native and pollution-tolerant species in streams in Santa Clara, California (EOA, Inc., 2001).

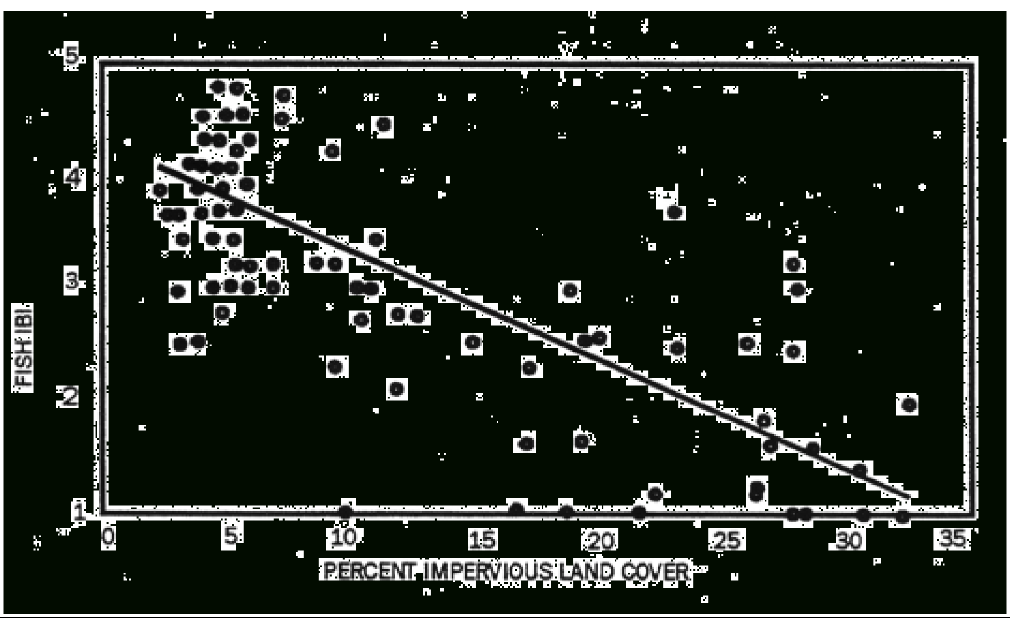


Figure 48: Fish-IBI vs. Watershed IC for Streams in the Patapsco River Basin, MD (Dail *et al.*, 1998)

Table 51: Recent Research Examining the Relationship Between Watershed IC and the Fish Community			
Biotic	**Key Finding (s)**	**Source**	**Location**
Abundance	Brown trout abundance and recruitment declined sharply at 10-15% IC.	Galli, 1994	MD
Salmonids	Seattle study showed marked reduction in coho salmon populations noted at 10-15% IC at nine streams.	Steward, 1983	WA
Anadromous Fish Eggs	Resident and anadromous fish eggs and larvae declined in 16 subwatersheds draining to the Hudson River with >10% IC area.	Limburg and Schmidt, 1990	NY
Community Index	1st, 2nd, and 3rd order streams in the Patapsco River Basin showed negative relationship between IBI and IC.	Dail *et al.*, 1998	MD
Community Index	Fish IBI and habitat scores were all ranked as poor in five subwatersheds that were greater than 30% IC.	Black and Veatch, 1994	MD
Community Index	In the Potomac subregion, subwatersheds with < 12% IC generally had streams in good to excellent condition based on a combined fish and aquatic insect IBI. Watersheds with >20% IC had streams in poor condition.	MNCPPC, 2000	MD
Community Index	In a two-year study of Piedmont streams draining eight watersheds representing various land uses in Chattahochee River Basin, fish community quality dropped sharply at an IC threshold of 15-30%.	Meyer and Couch, 2000	GA
Diversity	Of 23 headwater stream stations, all draining <10% IC areas, rated as good to fair; all with >12% were rated as poor. Fish diversity declined sharply with increasing IC between 10-12%.	Schueler and Galli, 1992	MD
Diversity, Sensitive Species	Comparison of 4 similar subwatersheds in Piedmont streams, there was significant decline in the diversity of fish at 10% IC. Sensitive species (trout and sculpin) were lost at 10-12%.	MWCOG, 1992	MD
Diversity, Community Index	In a comparison of watershed land use and fish community data for 47 streams between the 1970s and 1990s, a strong negative correlation was found between number species and IBI scores with effective connected IC. A threshold of 10% IC was observed with community quality highly variable below 10% but consistently low above 10% IC.	Wang *et al.*, 1997	WI
Diversity	In several dozen Piedmont headwater streams fish diversity declined significantly in areas beyond 10-12% IC.	Klein, 1979	MD
Diversity , Abundance, Non-native Species	IC strongly associated with several fisheries species and individual-level metrics, including number of pollution-tolerant species, diseased individuals, native and non-native species and total species present	EOA, Inc., 2001	CA
Juvenile Salmon Ratios	In Puget Sound study, the steepest decline of biological functioning was observed after six percent IC. There was a steady decline, with approximately 50% reduction in initial biotic integrity at 45% IC area.	Horner *et al.*, 1997	WA
Juvenile Salmon Ratio	Physical and biological stream indicators declined most rapidly during the initial phase of the urbanization process as total IC area exceeded the five to 10% range.	May *et al.*, 1997	WA
Salmonoid	Negative effects of urbanization (IC) with the defacto loss of non-structural BMPs (wetland forest cover and riparian integrity) on salmon ratios	Horner *et al.*, 2001	WA, MD, TX
Salmonoid, Sensitive Species	While no specific threshold was observed (impacts seen at even low levels of IC), Coho/cutthroat salmon ratios >2:1 were found when IC was < 5%. Ratios fell below one at IC levels below 20 %.	Horner and May, 1999	WA
Sensitive species, Salmonid	Three years stream sampling across the state (approximately 1000 sites), MBSS found that when IC was >15%, stream health was never rated good based on CBI, and pollution sensitive brook trout were never found in streams with >2% IC.	Boward *et al.*, 1999	MD
Sensitive Species, Salmonids	Seattle study observed shift from less tolerant coho salmon to more tolerant cutthroat trout population between 10 and 15% IC at nine sites.	Luchetti and Feurstenburg 1993	WA

Sensitive fish are defined as species that strongly depend on clean and stable bottom substrates for feeding and/or spawning. Sensitive fish often show a precipitous decline in urban streams. The loss of sensitive fish species and a shift in community structure towards more pollution-tolerant species is confirmed by multiple studies. Figure 50 shows the results of a comparison of four similar subwatersheds in the Maryland Piedmont that were sampled for the number of fish species present (MWCOG, 1992). As the level of watershed IC increased, the number of fish species collected dropped. Two sensitive species, including sculpin, were lost when IC increased from 10 to 12%, and four more species were lost when IC reached 25%. Significantly, only two species remained in the fish community at 55% watershed IC.

Salmonid fish species (trout and salmon) and anadromous fish species appear to be particularly impacted by watershed IC. In a study in the Pacific Northwest, sensitive coho salmon were seldom found in watersheds above 10 or 15% IC (Luchetti and Feurstenburg, 1993 and Steward, 1983). Key stressors in urban streams, such as higher peak flows, lower dry weather flows, and reduction in habitat complexity (e.g. fewer pools, LWD, and hiding places) are believed to change salmon species composition, favoring cutthroat trout populations over the natural coho populations (WDFW, 1997).

A series of studies from the Puget Sound reported changes in the coho/cutthroat ratios of juvenile salmon as watershed IC increased (Figure 51). Horner *et al.* (1999) found Coho/Cutthroat ratios greater than 2:1 in watersheds with less than 5 % IC. Ratios fell below 1:1 when IC exceeded 20%. Similar results were reported by May *et al.* (1997). In the mid-Atlantic region, native trout have stringent temperature and habitat requirements and are seldom present in watersheds where IC exceeds 15% (Schueler, 1994a). Declines in trout spawning success are evident above 10% IC. In a study of over 1,000 Maryland streams, Boward *et al.* (1999) found that sensitive brook trout were never found in streams that had more than 4% IC in their contributing watersheds.

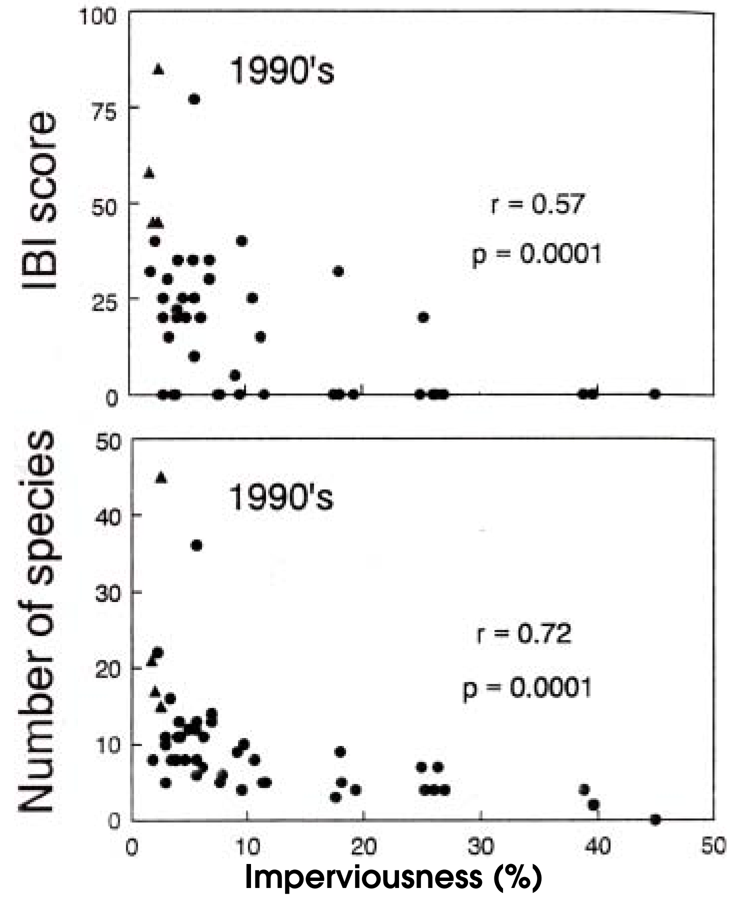


Figure 49: Fish-IBI and Number of Species vs. % IC in Wisconsin Streams (Wang *et al.*, 1997)

Fish Diversity

Anacostia River Basin

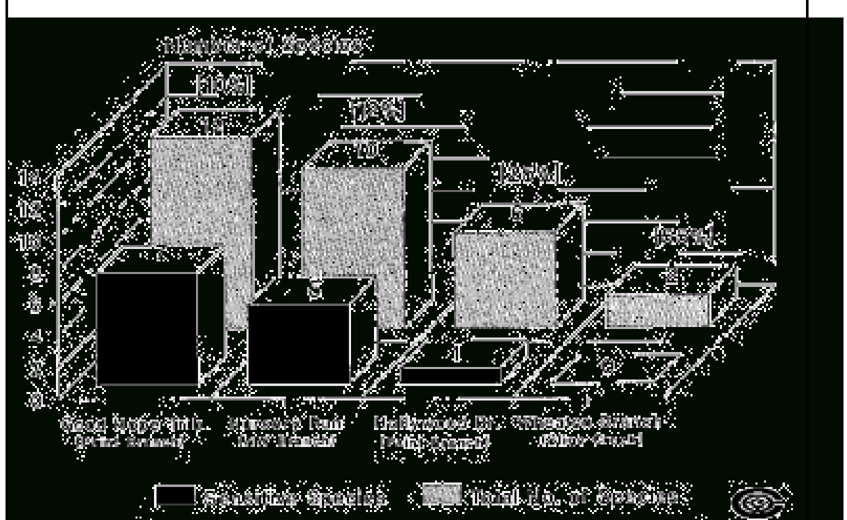


Figure 50: IC and Effects on Fish Species Diversity in Four Maryland Subwatersheds (MWCOG, 1992)

Table 52: Recent Research Examining Urbanization and Freshwater Fish Community Indicators

Biotic	Key Finding (s)	Source	Location
	Urbanization		
Community Index	All 40 urban sites sampled had fair to very poor IBI scores, compared to undeveloped reference sites.	Yoder, 1991	OH
Community Index	Negative correlations between biotic community and riparian conditions and forested areas were found. Similar levels of fish degradation were found between suburban and agricultural; urban areas were the most severe.	Meyer and Couch, 2000	GA
Community Index	Residential urban land use caused significant decrease in fish-IBI scores at 33%. In more urbanized Cuyahoga, a significant drop in IBI scores occurred around 8% urban land use in the watershed. When watersheds smaller than 100mi^2 were analyzed separately, the level of urban land associated with a significant drop in IBI scores occurred at around 15%. Above one du/ac, most sites failed to attain biocriteria regardless of degree of urbanization.	Yoder *et al.*, 1999	OH
Community Index, Abundance	As watershed development increased to about 10%, fish communities simplified to more habitat and trophic generalists and fish abundance and species richness declined. IBI scores for the urbanized stream fell from the good to fair category.	Weaver, 1991	VA
Diversity	A study of five urban streams found that as land use shifted from rural to urban, fish diversity decreased.	Masterson and Bannerman, 1994	WI
Diversity, Community Index	A comparison of three stream types found urban streams had lowest diversity and richness. Urban streams had IBI scores in the poor range.	Crawford and Lenat, 1989	NC
Salmon Spawning, Flooding Frequency	In comparing three streams over a 25-year period (two urbanizing and one remaining forested), increases in flooding frequencies and decreased trends in salmon spawning were observed in the two urbanizing streams, while no changes in flooding or spawning were seen in the forested system.	Moscript and Montgomery, 1997	WA
Sensitive Species	Observed dramatic changes in fish communities in most urbanizing stream sections, such as absence of brown trout and abundance of pollution-tolerant species in urban reaches.	Kemp and Spotila, 1997	PA
Sensitive Species, Diversity	Decline in sensitive species diversity and composition and changes in trophic structure from specialized feeders to generalists was seen in an urbanizing watershed from 1958 to 1990. Low intensity development was found to affect warm water stream fish communities similarly as more intense development.	Weaver and Garman, 1994	VA
Warm Water Habitat Biocriteria	25-30% urban land use defined as the upper threshold where attainment of warm water habitat biocriterion is effectively lost. Non-attainment also may occur at lower thresholds given the co-occurrence of stressors, such as pollution legacy, WTPs and CSOs.	Yoder and Miltner, 2000	OH
Community Index, Habitat	The amount of urban land use upstream of sample sites had a strong negative relationship with biotic integrity, and there appeared to be a threshold between 10 and 20% urban land use where IBI scores declined dramatically. Watersheds above 20% urban land invariably had scores less than 30 (poor to very poor). Habitat scores were not tightly correlated with degraded fish community attributes.	Wang *et al.*, 1997	WI
Community Index	A study in the Patapsco Basin found significant correlation of fish IBI scores with percent urbanized land over all scales (catchment, riparian area, and local area).	Roth *et al.*, 1998	MD

Table 52 (continued): Recent Research Examining Urbanization and Freshwater Fish Community Indicators			
Biotic	**Key Finding (s)**	**Source**	**Location**
Urbanization			
Sensitive Species	Evaluated effects of runoff in both urban and non-urban streams; found that native species dominated the non-urban portion of the watershed but accounted for only seven percent of species found in the urban portions of the watershed.	Pitt, 1982	CA
Other Land Use Indicators			
Community Index, Habitat	Atlanta study found that as watershed population density increased, there was a negative impact on urban fish and habitat. Urban stream IBI scores were inversely related to watershed population density, and once density exceeded four persons/acre, urban streams were consistently rated as very poor.	Couch *et al.*, 1997	GA
Community Index	In an Atlanta stream study, modified IBI scores declined once watershed population density exceeds four persons/acre in 21 urban watersheds	DeVivo *et al.*, 1997	GA
Community Index	In a six-county study (including Chicago, its suburbs and outlying rural/agricultural areas), streams showed a strong correlation between population density and fish community assessments such that as population density increased, community assessment scores went from the better - good range to fair - poor. Significant impacts seen at 1.5 people/acre.	Dreher, 1997	IL
Community Index	Similarly, negative correlations between biotic community and riparian conditions and forested areas were also found. Similar levels of fish degradation were found between suburban and agricultural; urban areas were the most severe.	Meyer and Couch, 2000	GA
Community Index	Amount of_forested land in basin directly related to IBI scores for fish community condition.	Roth *et al.*, 1996	MD
Salmonid, Sensitive Species	Species community changes from natural coho salmon to cutthroat trout population with increases in peak flow, lower low flow, and reductions in stream complexity.	WDFW, 1997	WA

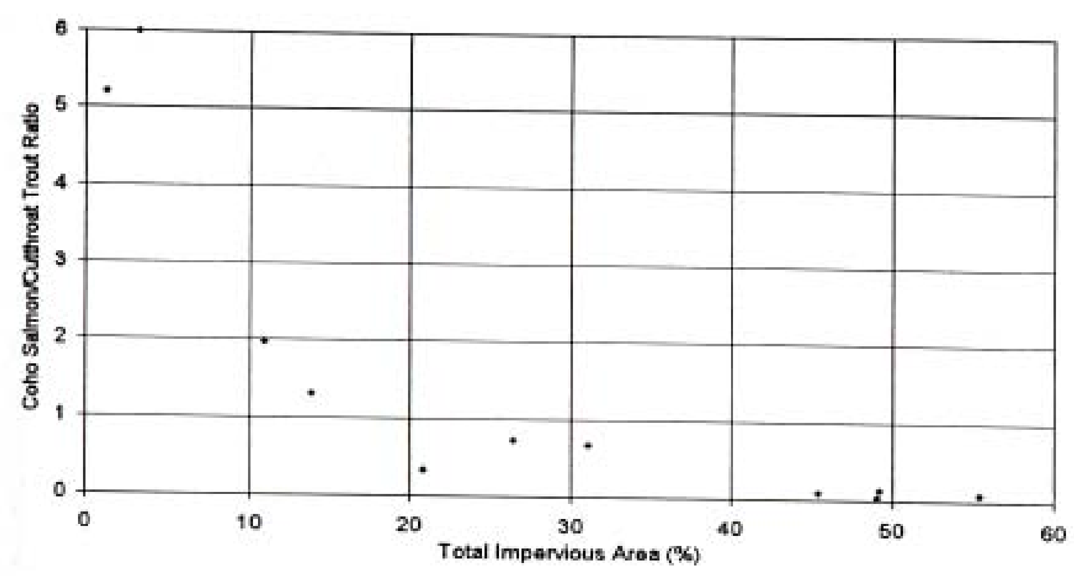


Figure 51: Coho Salmon/Cutthroat Trout Ratio for Puget Sound Streams (Horner *et al.*, 1997)

Many fish species have poor spawning success in urban streams and poor survival of fish eggs and fry. Fish barriers, low intragravel dissolved oxygen, sediment deposition and scour are all factors that can diminish the ability of fish species to successfully reproduce. For example, Limburg and Schmidt (1990) discovered that the density of anadromous fish eggs and larvae declined sharply in subwatersheds with more than 10% IC.

5.4.2 Findings Based on Other Development Indicators

Urban land use has frequently been used as a development indicator to evaluate the impact on fish diversity. Streams in urban watersheds typically had lower fish species diversity and richness than streams located in less developed watersheds. Declines in fish diversity as a function of urban land cover have been documented in numerous studies (Crawford and Lenat, 1989; Masterson and Bannerman, 1994; Roth *et al., 1998;* Yoder, 1991, and Yoder *et al.*, 1999). USEPA (1982) found that native fish species dominated the fish community of non-urban streams, but accounted for only 7% of the fish community found in urban streams. Kemp and Spotila (1997) evaluated streams in Pennsylvania and noted the loss of sensitive species (e.g. brown trout) and the increase of pollution-tolerant species, such as sunfish and creek chub (Figure 52).

Wang *et al.* (1997) cited percentage of urban land in Wisconsin watersheds as a strong negative factor influencing fish-IBI scores in streams and observed strong declines in IBI scores with 10 to 20% urban land use. Weaver and Garman (1994) compared the historical changes in the warm-water fish community of a Virginia stream that had undergone significant urbanization and found that many of the sensitive species present in 1958 were either absent or had dropped sharply in abundance when the watershed was sampled in 1990. Overall abundance had dropped from 2,056 fish collected in 1958 to 417 in 1990. In addition, the 1990 study showed that 67% of the catch was bluegill and common shiner, two species that are habitat and trophic "generalists." This shift in community to more habitat and trophic generalists was observed at 10% urban land use (Weaver, 1991).

Yoder *et al.* (1999) evaluated a series of streams in Ohio and reported a strong decrease in warm-water fish community scores around 33% residential urban land use. In the more urbanized Cuyahoga streams, sharp drops in

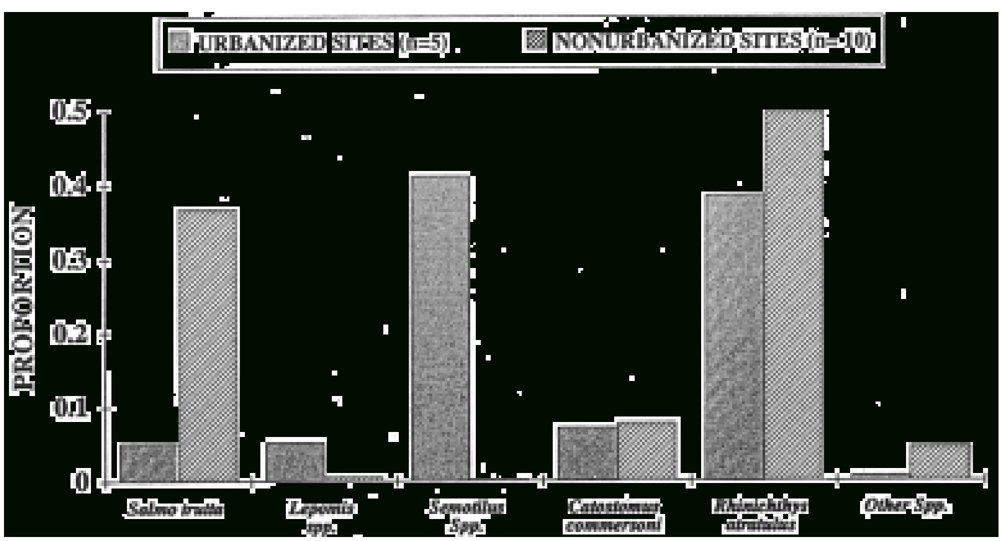


Figure 52: Mean Proportion of Fish Taxa in Urban and Non-Urban Streams, Valley Forge Watershed, PA (Kemp and Spotila, 1997)

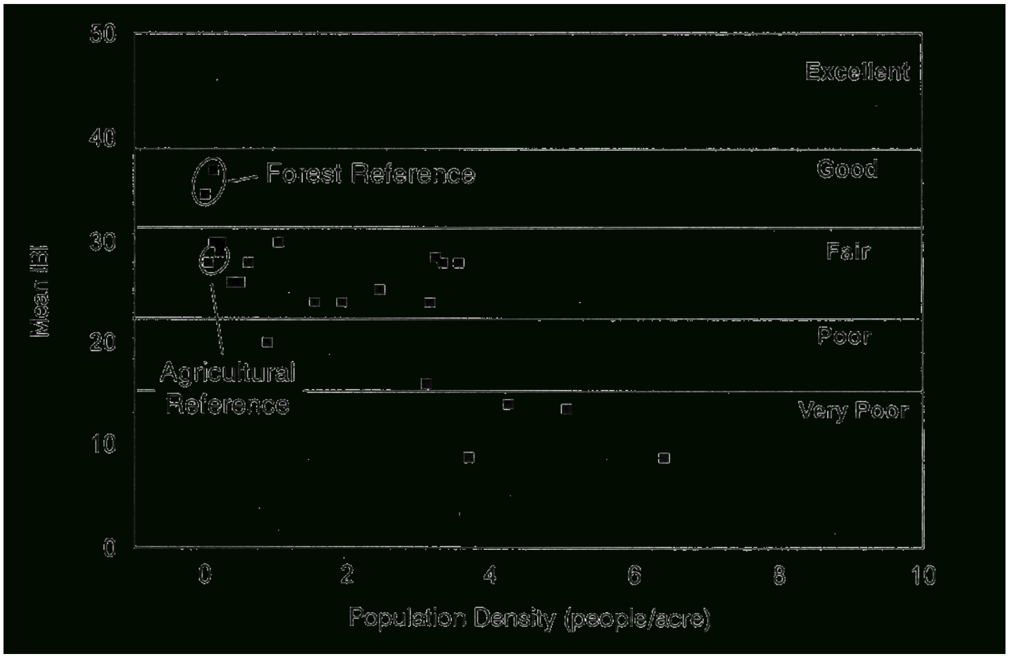


Figure 53: Relationship Between Watershed Population Density and Stream IBI Scores in Georgia Streams (DeVivo *et al.*, 1997)

fish-IBI scores occurred around 8% urban land use, primarily due to certain stressors which functioned to lower the non-attainment threshold. When watersheds smaller than 100mi^2 were analyzed separately, the percentage of urban land use associated with a sharp drop in fish-IBI scores was around 15%. In a later study, Yoder and Miltner (2000) described an upper threshold for quality warm-water fish habitat at 25 to 30% urban land use.

Watershed population and housing density have also been used as indicators of the health of the fish community. In a study of 21 urban watersheds in Atlanta, DeVivo *et al.* (1997) observed a shift in mean fish-IBI scores from "good to fair" to "very poor" when watershed population density exceeded four people/acre (Figure 53). A study of Midwest streams in metropolitan Illinois also found a negative relationship between increase in population density and fish communities, with significant impacts detected at population densities of 1.5 people or greater per acre (Dreher, 1997). In the Columbus and Cuyahoga watersheds in Ohio, Yoder *et al*. (1999) concluded that most streams failed to attain fish biocriteria above one dwelling unit/acre.

5.5 Effects on Amphibian Diversity

Amphibians spend portions of their life cycle in aquatic systems and are frequently found within riparian, wetland or littoral areas. Relatively little research has been conducted to directly quantify the effects of watershed development on amphibian diversity. Intuitively, it would appear that the same stressors that affect fish and aquatic insects would also affect amphibian species, along with riparian wetland alteration. We located four research studies on the impacts of watershed urbanization on amphibian populations; only one was related to streams (Boward *et al.,* 1999), while others were related to wetlands (Table 53).

A primary factor influencing amphibian diversity appears to be water level fluctuations (WLF) in urban wetlands that occur as a result of increased stormwater discharges. Chin (1996) hypothesized that increased WLF and other hydrologic factors affected the abundance of egg clutches and available amphibian breeding habitat, thereby ultimately influencing amphibian richness. Increased WLF can limit reproductive success by eliminating mating habitat and the emergent vegetation to which amphibians attach their eggs.

Taylor (1993) examined the effect of watershed development on 19 freshwater wetlands in King County, WA and concluded that the additional stormwater contributed to greater annual WLF. When annual WLF exceeded about eight inches, the richness of both the wetland plant and amphibian communities dropped sharply. Large increases in WLF were consistently observed in freshwater wetlands when IC in upstream watersheds exceeded 10 to 15%. Further research on streams and wetlands in the Pacific northwest by Horner *et al.* (1997) demonstrated the correlation between watershed IC and diversity of amphibian species. Figure 54 illustrates the relationship between amphibian species abundance and watershed IC, as documented in the study.

Table 53: Recent Research on the Relationship Between Percent Watershed Urbanization and the Amphibian Community			
Indicator	**Key Finding(s)**	**Reference Year**	**Location**
% IC			
Reptile and Amphibian Abundance	In a three-year stream sampling across the state (approximately 1000 sites), MBSS found only hardy pollution-tolerant reptiles and amphibians in stream corridors with >25% IC drainage area.	Boward *et al.*, 1999	MD
Amphibian Density	Mean annual water fluctuation inversely correlated to amphibian density in urban wetlands. Declines noted beyond 10% IC.	Taylor, 1993	WA
Other Studies			
Species Richness	In 30 wetlands, species richness of reptiles and amphibians was significantly related to density of paved roads on lands within a two kilometer radius.	Findlay and Houlahan, 1997	Ontario
Species Richness	Decline in amphibian species richness as wetland WLF increased. While more of a continuous decline rather than a threshold, WLF = 22 centimeters may represent a tolerance boundary for amphibian community.	Horner *et al.*, 1997	WA
Amphibian Density	Mean annual water fluctuation inversely correlated to amphibian density in urban wetlands.	Taylor, 1993	WA

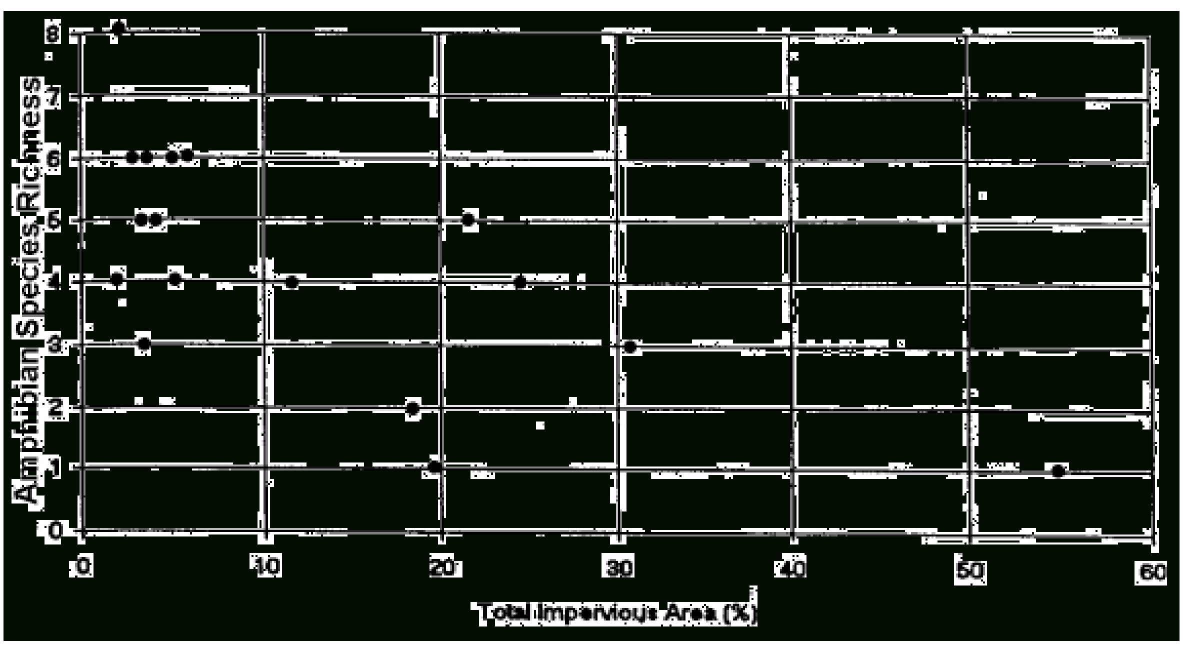


Figure 54: Amphibian Species Richness as a Function of Watershed IC in Puget Sound Lowland Wetlands (Horner *et al.*, 1997)

5.6 Effects on Wetland Diversity

We found a limited number of studies that evaluated the impact of watershed urbanization on wetland plant diversity (Table 54). Two studies used IC as an index of watershed development and observed reduced wetland plant diversity around or below 10% IC (Hicks and Larson, 1997 and Taylor, 1993). WLF and road density were also used as indicators (Findlay and Houlahan, 1997; Horner *et al.*, 1997; Taylor, 1993).

Horner *et al.* (1997) reported a decline in plant species richness in emergent and scrub-shrub wetland zones of the Puget Sound region as WLF increased. They cautioned that species numbers showed a continuous decline rather than a threshold value; however, it was indicated that WLF as small as 10 inches can represent a tolerance boundary for wetland plant communities. Horner further stated that in 90% of the cases where WLF exceeded 10 inches, watershed IC exceeded 21%.

Table 54: Recent Research Examining the Relationship Between Watershed Development and Urban Wetlands

Watershed Indicator / Biotic	Key Finding(s)	Reference	Location
% IC			
Insect Community	Significant declines in various indicators of wetland aquatic macro-invertebrate community health were observed as IC increased to 8-9%.	Hicks and Larson, 1997	CT
WLF, Water Quality	There is a significant increase in WLF, conductivity, fecal coliform bacteria, and total phosphorus in urban wetland as IC exceeds 3.5%.	Taylor *et al.*, 1995	WA
Plant Density	Declines in urban wetland plant density noted in areas beyond 10% IC.	Taylor, 1993	WA
Other Watershed Indicators			
Plant Density	Mean annual water fluctuation inversely correlated to plant density in urban wetlands.	Taylor, 1993	WA
Plant Species Richness	Decline in plant species richness in emergent and scrub-shrub wetland zones as WLF increased. While more of a continuous decline, rather than a threshold, WLF=22 centimeters may represent a tolerance boundary for the community	Horner *et al.*, 1997	WA
Plant Species Richness	In 30 wetlands, species richness was significantly related to density of paved roads within a two kilometer radius of the wetland. Model predicted that a road density of 2kilometers per hectare in paved road within 1000 meters of wetland will lead to a 13% decrease in wetland plant species richness.	Findlay and Houlahan,1997	Ontario

5.7 Effects on Freshwater Mussel Diversity

Freshwater mussels are excellent indicators of stream quality since they are filter-feeders and essentially immobile. The percentage of imperiled mussel species in freshwater ecoregions is high (Williams *et al.*, 1993). Of the 297 native mussel species in the United States, 72% are considered endangered, threatened, or of special concern, including 21 mussel species that are presumed to be extinct. Seventy mussel species (24%) are considered to have stable populations, although many of these have declined in abundance and distribution. Modification of aquatic habitats and sedimentation are the primary reasons cited for the decline of freshwater mussels (Williams *et al.,* 1993).

Freshwater mussels are very susceptible to smothering by sediment deposition. Consequently, increases in watershed development and sediment loading are suspected to be a factor leading to reduced mussel diversity. At sublethal levels, silt interferes with feeding and metabolism of mussels in general (Aldridge *et al.*, 1987). Major sources of mortality and loss of diversity in mussels include impoundment of rivers and streams, and eutrophication (Bauer, 1988). Changes in fish diversity and abundance due to dams and impoundments can also influence the availability of mussel hosts (Williams *et al.*, 1992).

Freshwater mussels are particularly sensitive to heavy metals and pesticides (Keller and Zam, 1991). Although the effects of metals and pesticides vary from one species to another, sub-lethal levels of PCBs, DDT, Malathion, Rotenone and other compounds are generally known to inhibit respiratory efficiency and accumulate in tissues (Watters, 1996). Mussels are more sensitive to pesticides than many other animals tested and often act as “first-alerts” to toxicity long before they are seen in other organisms.

We were unable to find any empirical studies relating impacts of IC on the freshwater mussel communities of streams.

5.8 Conclusion

The scientific record is quite strong with respect to the impact of watershed urbanization on the integrity and diversity of aquatic communities. We reviewed 35 studies that indicated that increased watershed development led to declines in aquatic insect diversity and about 30 studies showing a similar impact on fish diversity. The scientific literature generally shows that aquatic insect and freshwater fish diversity declines at fairly low levels of IC (10 to 15%), urban land use (33%), population density (1.5 to eight people/acre) and housing density (>1 du/ac). Many studies also suggest that sensitive elements of the aquatic community are affected at even lower levels of IC. Other impacts include loss of sensitive species and reduced abundance and spawning success. Research supports the ICM, although additional research is needed to establish the upper threshold at which watershed development aquatic biodiversity can be restored.

One area where more research is needed involves determining how regional and climatic variations affect aquatic diversity in the ICM. Generally, it appears that the 10% IC threshold applies to streams in the East Coast and Midwest, with Pacific Northwest streams showing impacts at a slightly higher level. For streams in the arid and semi-arid Southwest, it is unclear what, if any, IC threshold exists given the naturally stressful conditions for these intermittent and ephemeral streams (Maxted, 1999). Southwestern streams are characterized by seasonal bursts of short but intense rainfall and tend to have aquatic communities that are trophically simple and relatively low in species richness (Poff and Ward, 1989).

Overall, the following conclusions can be drawn:

- IC is the most commonly used index to assess the impacts of watershed urbanization on aquatic insect and fish diversity. Percent urban land use is also a common index.

- The ICM may not be sensitive enough to predict biological diversity in watersheds with low IC. For example, below 10% watershed IC, other watershed variables such as riparian continuity, natural forest cover, cropland, ditching and acid rain may be better for predicting stream health.

- More research needs to be done to determine the maximum level of watershed development at which stream diversity can be restored or maintained. Additionally, the capacity of stormwater treatment practices and stream buffers to mitigate high levels of watershed IC warrants more systematic research.

- More research is needed to test the ICM on amphibian and freshwater mussel diversity.

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Glossary

1st order stream: The smallest perennial stream. A stream that carries water throughout the year and does not have permanently flowing tributaries.

2nd order stream: Stream formed by the confluence of two 1st order streams.

3rd order stream: Stream formed by the confluence of two 2nd order streams.

Acute toxicity: Designates exposure to a dangerous substance or chemical with sufficient dosage to precipitate a severe reaction, such as death.

Alluvial: Pertaining to processes or materials associated with transportation or deposition by running water.

Anadromous: Organisms that spawn in freshwater streams but live most of their lives in the ocean.

Annual Pollutant Load: The total mass of a pollutant delivered to a receiving water body in a year.

Bankfull: The condition where streamflow just fills a stream channel up to the top of the bank and at a point where the water begins to overflow onto a floodplain.

Baseflow: Stream discharge derived from ground water that supports flow in dry weather.

Bedload: Material that moves along the stream bottom surface, as opposed to suspended particles.

Benthic Community: Community of organisms living in or on bottom substrates in aquatic habitats, such as streams.

Biological Indicators: A living organism that denotes the presence of a specific environmental condition.

Biological Oxygen Demand (BOD): An indirect measure of the concentration of biologically degradable material present in organic wastes. It usually reflects the amount of oxygen consumed in five days by bacterial processes breaking down organic waste.

Carcinogen: A cancer-causing substance or agent.

Catchment: The smallest watershed management unit. Defined as the area of a development site to its first intersection with a stream, usually as a pipe or open channel outfall.

Chemical Oxygen Demand (COD): A chemical measure of the amount of organic substances in water or wastewater. Non-biodegradable and slowly degrading compounds that are not detected by BOD are included.

Chronic Toxicity: Showing effects only over a long period of time.

Combined Sewer Overflow (CSO): Excess flow (combined wastewater and stormwater runoff) discharged to a receiving water body from a combined sewer network when the capacity of the sewer network and/or treatment plant is exceeded, typically during storm events.

Combined Indices (C-IBI or CSPS): Combined indices that use both fish and aquatic insect metrics and a variety of specific habitat scores to classify streams.

Cryptosporidium parvum: A parasite often found in the intestines of livestock which contaminates water when animal feces interacts with a water source.

Deicer: A compound, such as ethylene glycol, used to melt or prevent the formation of ice.

Dissolved Metals: The amount of trace metals dissolved in water.

Dissolved Phosphorus: The amount of phosphorus dissolved in water.

Diversity: A numerical expression of the evenness and distribution of organisms.

Ecoregion: A continuous geographic area over which the climate is uniform to permit the development of similar ecosystems on sites with similar geophysical properties.

Embeddedness: Packing of pebbles or cobbles with fine-grained silts and clays.

EPT Index: A count of the number of families of each of the three generally pollution-sensitive orders: Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies).

Escherichia coli (E. coli): A bacteria that inhabits the intestinal tract of humans and other warm-blooded animals. Although it poses no threat to human health, its presence in drinking water does indicate the presence of other, more dangerous bacteria.

Eutrophication: The process of over-enrichment of water bodies by nutrients, often typified by the presence of algal blooms.

Fecal coliform: Applied to E. coli and similar bacteria that are found in the intestinal tract of humans and animals. Coliform bacteria are commonly used as indicators of the presence of pathogenic organisms. Their presence in water indicates fecal pollution and potential contamination by pathogens.

Fecal streptococci: Bacteria found in the intestine of warm-blooded animals. Their presence in water is considered to verify fecal pollution.

Fish Blockages: Infrastructures associated with urbanization, such as bridges, dams, and culverts, that affect the ability of fish to move freely upstream and downstream in watersheds. Can prevent re-colonization of resident fish and block the migration of anadromous fish.

Flashiness: Percent of flows exceeding the mean flow for the year. A flashy hydrograph would have larger, shorter-duration hydrograph peaks.

Geomorphic: The general characteristic of a land surface and the changes that take place in the evolution of land forms.

Giardia lamblia: A flagellate protozoan that causes severe gastrointestinal illness when it contaminates drinking water.

Herbicide: Chemicals developed to control or eradicate plants.

Hotspot: Area where land use or activities generate highly contaminated runoff, with concentrations of pollutants in excess of those typically found in stormwater.

Hydrograph: A graph showing variation in stage (depth) or discharge of a stream of water over a period of time.

Illicit discharge: Any discharge to a municipal separate storm sewer system that is not composed entirely of storm water, except for discharges allowed under an NPDES permit.

Impervious Cover: Any surface in the urban landscape that cannot effectively absorb or infiltrate rainfall.

Impervious Cover Model (ICM): A general watershed planning model that uses percent watershed impervious cover to predict various stream quality indicators. It predicts expected stream quality declines when watershed IC exceeds 10% and severe degradation beyond 25% IC.

Incision: Stream down-cuts and the channel expands in the vertical direction.

Index of Biological Integrity (IBI): Tool for assessing the effects of runoff on the quality of the aquatic ecosystem by comparing the condition of multiple groups of organisms or taxa against the levels expected in a healthy stream.

Infiltration: The downward movement of water from the surface to the subsoil. The infiltration capacity is expressed in terms of inches per hour.

Insecticide: Chemicals developed to control or eradicate insects.

Large Woody Debris (LWD): Fundamental to stream habitat structure. Can form dams and pools; trap sediment and detritus; provide stabilization to stream channels; dissipate flow energy and promote habitat complexity.

Mannings N: A commonly used roughness coefficient; actor in velocity and discharge formulas representing the effect of channel roughness on energy losses in flowing water.

Methyl Tertiary-Butyl Ether: An oxygenate and gasoline additive used to improve the efficiency of combustion engines in order to enhance air quality and meet air pollution standards. MTBE has been found to mix and move more easily in water than many other fuel components, thereby making it harder to control, particularly once it has entered surface or ground waters.

Microbe: Short for microorganism. Small organisms that can be seen only with the aid of a microscope. Most frequently used to refer to bacteria. Microbes are important in the degradation and decomposition of organic materials.

Nitrate: A chemical compound having the formula NO_3. Excess nitrate in surface waters can lead to excessive growth of aquatic plants.

Organic Matter: Plant and animal residues, or substances made by living organisms. All are based upon carbon compounds.

Organic Nitrogen: Nitrogen that is bound to carbon-containing compounds. This form of nitrogen must be subjected to mineralization or decomposition before it can be used by the plant community.

Overbank Flow: Water flow over the top of the bankfull channel and onto the floodplain.

Oxygenate: To treat, combine, or infuse with oxygen.

Peak Discharge: The maximum instantaneous rate of flow during a storm, usually in reference to a specific design storm event.

Pesticides: Any chemical agent used to control specific organisms, for example, insecticides, herbicides, fungicides and rodenticides.

Piedmont: Any plain, zone or feature located at the foot of a mountain. In the United States, the Piedmont (region) is a plateau extending from New Jersey to Alabama and lying east of the Appalachian Mountains.

Pool: A stream feature where there is a region of deeper, slow-moving water with fine bottom materials. Pools are the slowest and least turbulent of the riffle/run/pool category.

Protozoan: Any of a group of single-celled organisms.

Rapid Bioassessment Protocols (RBP): An integrated assessment, comparing habitat, water quality and biological measures with empirically defined reference conditions.

Receiving Waters: Rivers, lakes, oceans, or other bodies of water that receive water from another source.

Riffle: Shallow rocky banks in streams where water flows over and around rocks disturbing the water surface; often associated with whitewater. Riffles often support diverse biological communities due to their habitat niches and increased oxygen levels created by the water disturbance. Riffles are the most swift and turbulent in the riffle/run/pool category.

Roughness: A measurement of the resistance that streambed materials, vegetation, and other physical components contribute to the flow of water in the stream channel and flood-plain. It is commonly measured as the Manning's roughness coefficient (Manning's N).

Run: Stream feature characterized by water flow that is moderately swift flow, yet not particularly turbulent. Runs are considered intermediate in the riffle/run/pool category.

Runoff Coefficient: A value derived from a site impervious cover value that is applied to a given rainfall volume to yield a corresponding runoff volume.

Salmonid: Belonging to the family Salmonidae, which includes trout and salmon.

Sanitary Sewer Overflow (SSO): Excess flow of wastewater (sewage) discharged to a receiving water body when the capacity of the sewer network and/or treatment plant is exceeded, typically during storm events.

Semi-arid: Characterized by a small amount of annual precipitation, generally between 10 and 20 inches.

Simple Method: Technique used to estimate pollutant loads based on the amount of IC found in a catchment or subwatershed.

Sinuosity: A measure of channel curvature, usually quantified as the ratio of the length of the channel to the length of a straight line along the valley axis. It is, in essence, a ratio of the stream's actual running length to its down-gradient length.

Soluble Phosphorus: The amount of phosphorus available for uptake by plants and animals.

Stormwater: The water produced as a result of a storm.

Subwatershed: A smaller geographic section of a larger watershed unit with a drainage area of between two to 15 square miles and whose boundaries include all the land area draining to a point where two 2nd order streams combine to form a 3rd order stream.

Total Dissolved Solids (TDS): A measure of the amount of material dissolved in water (mostly inorganic salts).

Total Kjeldhal Nitrogen (TKN): The total concentration of nitrogen in a sample present as ammonia or bound in organic compounds.

Total Recoverable Metals: The amount of a metal that is in solution after a representative suspended sediment sample has been digested by a method (usually using a dilute acid solution) that results in dissolution of only readily soluble substances).

Total Maximum Daily Load (TMDL): The maximum quantity of a particular water pollutant that can be discharged into a body of water without violating a water quality standard.

Total Nitrogen (Total N): A measure of the total amount of nitrate, nitrite and ammonia concentrations in a body of water.

Total Organic Carbon (TOC): A measure of the amount of organic material suspended or dissolved in water.

Total Phosphorous (Total P): A measure of the concentration of phosphorus contained in a body of water.

Total Suspended Solids (TSS): The total amount of particulate matter suspended in the water column.

Trophic Level: The position of an organism in a food chain or food pyramid.

Turbidity: A measure of the reduced transparency of water due to suspended material which carries water quality and aesthetic implications. Applied to waters containing suspended matter that interferes with the passage of light through the water or in which visual depth is restricted.

Volatile Organic Compounds (VOC): Chemical compounds which are easily transported into air and water. Most are industrial chemicals and solvents. Due to their low water solubility they are commonly found in soil and water.