D. Hoffman, B. Rattner, G.A. Burton, Jr., and J.Cairns, Jr. *Handbook of Ecotoxicology, 2nd Edition*. CRC-Lewis. Boca Raton, FL, 2002.

Receiving Water Impacts Associated with Urban Runoff

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Introduction

The main purpose of treating stormwater is to reduce its adverse impacts on receiving water beneficial uses. Therefore, it is important in any urban stormwater study to assess the detrimental effects that runoff is actually having on a receiving water. Urban receiving waters may have many beneficial use goals, including:

- stormwater conveyance (flood prevention)
- biological uses (warm water fishery, biological integrity, etc.)
- non-contact recreation (linear parks, aesthetics, boating, etc.)
- contact recreation (swimming)
- water supply

Two joint research projects recently funded by the EPA[145, 146] examined the historical development of stormwater management programs and modifications that should be incorporated into future design procedures. The projects found that with full development in an urban watershed and with no stormwater controls, it is unlikely that any of the above listed uses can be fully obtained. With less development and with the application of stormwater controls, some uses may be possible. It is important that unreasonable expectations not be placed on urban waters, as the cost to obtain these uses may be prohibitive. With full-scale development and lack of adequate stormwater controls, severely degraded streams will be common. However, stormwater conveyance and aesthetics should be the basic beneficial use goals for all urban waters. Biological integrity should also be a goal, but with the realization that the natural stream ecosystem will be severely modified with urbanization. Certain basic controls, installed at the time of development, plus protection of stream habitat, may enable partial use of some of these basic goals in urbanized watersheds. Careful planning and optimal utilization of stormwater controls are necessary to obtain these basic goals in most watersheds. Water contact recreation, consumptive fisheries, and water supplies are not appropriate goals for most urbanized watersheds. However, these higher uses may be possible in urban areas where the receiving waters are large and drain mostly undeveloped areas.

Water Environment & Technology[1] reported that the latest National Water Quality Inventory released by the EPA only showed a slight improvement in the attainment of beneficial uses in the nations waters. Urban runoff was cited as the leading source of problems in estuaries, with nutrients and bacteria as the leading problems. Problems in rivers and lakes were mostly caused by agricultural runoff, with urban runoff the third ranked source for lakes, and the fourth ranked source for rivers. Bacteria, siltation, and nutrients were the leading problems in the nations rivers and lakes. Borchardt and Statzner[2] stressed that many conditions may affect receiving waters from stormwater, specifically physical factors (such as shear stress) and chemical factors (such as oxygen depletion and/or non-ionized ammonia).

In general, monitoring of urban stormwater runoff has indicated that the biological beneficial uses of urban receiving waters are most likely affected by habitat destruction and long-term exposures to contaminants (especially to macroinvertebrates via contaminated sediment), while documented effects associated from acute exposures of toxicants in the water column are rare[3-5]. Receiving water contaminant concentrations resulting from runoff events and typical laboratory bioassay test results have not indicated many significant short-term receiving water problems. As an example, Lee and Jones-Lee[6] state that exceedences of numeric criteria by short-term discharges do not necessarily imply that a beneficial use impairment exists. Many toxicologists and water quality expects have concluded that the relatively short periods of exposures to the toxicant concentrations in stormwater are not sufficient to produce the receiving water effects that are evident in urban receiving waters, especially considering the relatively large portion of the toxicants that are associated with particulates[7]. Lee and Jones-Lee[7] conclude that the biological problems evident in urban receiving waters are mostly associated with illegal discharges and that the sediment bound toxicants are of little risk. Mancini and Plummer[8] have long been advocates of numeric water quality standards for stormwater that reflect the partitioning of the toxicants and the short periods of exposure during rains. Unfortunately, this approach attempts to isolate individual runoff events and does not consider the accumulative adverse effects caused by the frequent exposures of receiving water organisms to stormwater[9-11]. Recent investigations have identified acute toxicity problems associated with moderate-term (about 10 to 20 day) exposures to adverse toxicant concentrations in urban receiving streams[12]. However, the most severe receiving water problems are likely associated with chronic exposures to contaminated sediment and to habitat destruction.

Pathogens in stormwater are also a significant concern potentially affecting human health. The use of indicator bacteria is controversial for stormwater, as well as the assumed time of typical exposure of swimmers to contaminated receiving waters. However, recent epidemiological studies has shown significant health effects associated with stormwater contaminated marine swimming areas. Protozoa pathogens, especially associated with likely sewage-contaminated stormwater, is also of public health concem.

Evaluating a receiving water and understanding the potential role that urban wet weather flows may have on its beneficial uses is a complex and time consuming activity. Burton and Pitt[13] have produced a comprehensive book describing the development of effective monitoring strategies, including selection of parameters, development of the experimental design, and detailed guidance on sampling, analyses, and data interpretation.

Urban runoff has been found to cause significant receiving water impacts on aquatic life[3,4,13]. The effects are obviously most severe for receiving waters draining heavily urbanized watersheds. However, some studies have shown important aquatic life impacts for streams in watersheds that are less than ten percent urbanized[19,22].

In order to best identify and understand these impacts, it is necessary to include biological monitoring, using a variety of techniques, and sediment quality analyses, in a monitoring program. Water column testing alone has been shown to be very misleading. Most aquatic life impacts associated with urbanization are probably related to long-term problems caused by polluted sediments and food web disruption. Transient water column quality conditions associated with urban runoff probably rarely cause significant aquatic life impacts.

The underlying theme of these researchers is that an adequate analysis of receiving water biological impacts must include investigations of a number of biological organism groups (fish, benthic macroinvertebrates, algae, rooted macrophytes, etc.) in addition to studies of water and sediment quality[13]. Simple studies of water quality alone, even with possible comparisons with water quality criteria for the protection of aquatic life, are usually inadequate to predict biological imp acts associated with urban runoff.

Duda, *et al*.[14] presented a discussion on why traditional approaches for assessing water quality, and selecting control options, in urban areas have failed. The main difficulties of traditional approaches when used with urban runoff are: the complexity of contaminant sources, wet weather monitoring problems, and limitations when using water quality standards to evaluate the severity of wet weather receiving water problems. They also discuss the difficulty of meeting water quality goals in urban areas that were promulgated in the Water Pollution Control Act.

Relationships between observed receiving water biological effects and possible causes have been especially difficult to identify, let alone quantify. The studies reported in this paper have identified a wide variety of possible causative agents, including sediment contamination, poor water quality (low dissolved oxygen, high toxicants, etc.), and factors effecting the physical habitat of the stream (high flows, unstable streambeds, absence of refuge areas, etc.). It is expected that all of these factors are problems, but their relative importance varies greatly depending on the watershed and receiving water conditions. Horner[15], as an example, notes that many watershed, site, and organism specific factors must be determined before the best combination of runoff control practices to protect aquatic life can be determined.

The time scale of biological impacts in receiving waters affected by stormwater must also be considered. Snodgrass, *et al*.[16] reported that ecological responses to watershed changes may take between 5 and 10 years to equilibrate. Therefore, receiving water investigations conducted soon after disturbances or mitigation may not accurately reflect the long-term conditions that will eventually occur. They found that the first changes due to urbanization will be to stream and groundwater hydrology, followed by fluvial morphology, then water quality, and finally the aquatic ecosystem. They also reported that it is not possible to predict biological responses from in-stream habitat changes or conditions, although they, along with many other researchers[26, 62-67, 77, 80-82, 85-88, 99, 102, 103, 105-108] have found that habitat changes are among the most serious causes of the aquatic biological problems associated with urbanization of a watershed.

Gross Indicators of Acute Aquatic Organism Stress in Urban Receiving Waters

Dissolved Oxygen Depletion Investigations

Dissolved oxygen stream levels have historically been used to indicate receiving water problems associated with point source contaminant discharges and with combined sewer overflows. Therefore, early investigations of the effects of stormwater discharges mostly focused on in-stream dissolved oxygen conditions downstream from outfalls. Of course, DO levels are also being evaluated in most current receiving water investigations also, but the emphasis has shifted more towards elevated nutrient and toxicant concentrations, plus numerous other indicators of aquatic organism stress, as described later.

An early study of DO in urban streams only affected by stormwater was conducted by Ketchum[17] in Indiana. Sampling was conducted at nine cities and the project was designed to detect significant dissolved oxygen deficits in streams during periods of rainfall and runoff. The results of this study indicated that wet weather DO levels generally appear to be similar or higher than those observed during dry weather conditions in the same streams. They found that significant wet weather DO depletions were not observed, and due to the screening nature of the sampling program, more subtle impacts could not be measured. Heaney, *et al*.[18], during their review of studies that examined continuous dissolved oxygen (DO) monitoring stations downstream from urbanized areas, indicated that the worst dissolved oxygen levels occurred after the storms in about one-third of the cases studied. This lowered DO could be due to urban runoff moving downstream, combined sewer overflows and/or resuspension of benthic deposits. Resuspended benthic deposits could have been previously settled urban runoff settleable solids. They also found that worst case conditions do not always occur during the low flow periods following storms. As noted below, adverse dissolved oxygen conditions associated with urban runoff are likely to occur a substantial time after the runoff event and downstream from the discharge locations.

Figure 1 illustrates a problem that may be common to DO predictions in urban receiving waters. Pitt[19] conducted three long-term BOD experiments with stormwater collected from a residential area in San Jose, CA. These were conventional BOD tests, using approved procedures published in the then current version of *Standard Methods*. Basically, many BOD bottles were prepared for each sample, representing replicates for each day for the observations, and for several different dilutions. The bottles were seeded with an activated sludge seed to provide a starting microbial population. As seen, the observed BOD curves do not have a conventional shape. The BOD_5 values are about 25 mg/L, typical to what is commonly reported for most stormwater. However, the BOD curves are seen to rapidly increase throughout the 20-day test period, instead of leveling off at about 7 to 10 days, as expected for municipal wastewaters. These curves illustrate the common problem of acclimation of a wastewater to the microorganisms that are present in the test solution. Stormwater has relatively low levels of nutrients and easily assimilated organic material, but moderate levels of toxicants. It is possible that the activated sludge seed requires extra time for the microbial population to shift to a population dominated by organisms capable of effectively degrading the organics in stormwater. Alternatively (or in addition), the more refractory organics in stormwater may simply require a longer period of time for degradation. In any case, the ultimate BOD/BOD_5 ratio for stormwater is much greater than for conventional municipal wastewaters, making simple use of observed BOD_5 values in receiving water models problematic. Urban stream sediments are commonly anaerobic, likely caused by the deposition of the slowly decaying stormwater organic compounds. Stormwater effects on short-term stream DO levels may be minimal, but sediment interaction (including scour) with the water can have adverse effects long after the stormwater event that discharged the decaying material. Therefore, the misuse of the classical BOD_5 test for stormwater can lead to poor conclusions concerning urban DO conditions, one of the more commonly used indicators of ecological health.

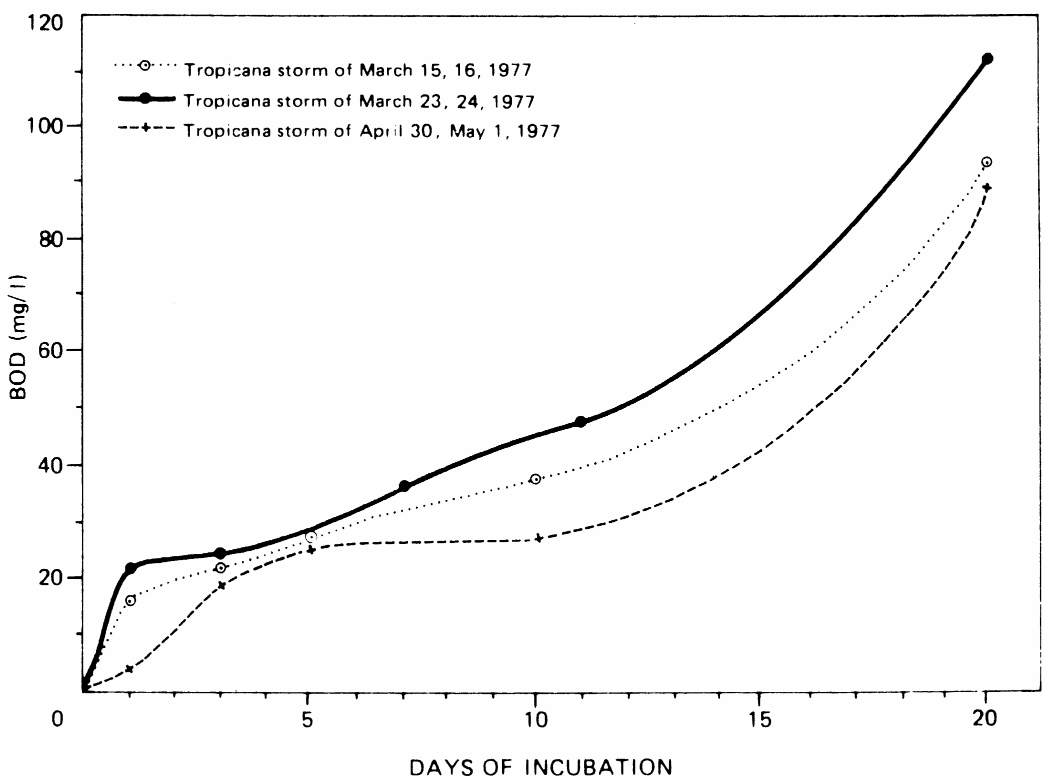

Figure 1. Long-term BOD tests for stormwater[19].

Urban Runoff Effects on Receiving Water Contaminant Concentrations

Numerous data are available characterizing stormwater chemical characteristics. This discussion summarizes a few example cases where in-stream measurements found significant changes in quality as a function of land use. These studies usually sampled streams as they passed through urban areas, from upstream relatively uncontaminated areas through and past urban areas. Both wet and dry weather sampling was also usually conducted.

In the southeast, many urban lakes in developing areas are typically characterized by high turbidity levels caused by high erosion rates of fine grained clays. There has been conflicting evidence on the role of these elevated turbidity levels on eutrophication processes and resulting highly fluctuating DO levels. Because of the high sediment loads, these urban lakes are quite different compared to most studied impoundments. Burkholder, *et al*.[20] described a series of enclosure experiments they conducted in Durant Reservoir, near Raleigh, North Carolina. The experimental design allowed investigating the effects of different levels of sediment and nutrients on algal productivity. They found that the effects (reduction of light reduction and coflocculation of clay and phosphate) of low (about 5 mg/L) and moderately high clay (about 15 mg/L) loadings added every 7 to 14 days did not significantly reduce the algal productivity simulation caused by high phosphate loadings. However, they noted that other investigators using higher clay loadings (about 25 mg/L added every 2 days) did see depressed effects of phosphorus enrichment on the test lake. They concluded that dynamically turbid systems, such as represented in southeastern urban lakes, have complex interacting mechanisms between discharged clay and nutrients that make simple predictions of the effects of eutrophication much more difficult than in the more commonly studied clear lakes. In general, they concluded that increased turbidity will either have no effect, or will have a mitigating effect, on the cultural eutrophication process.

Field and Cibik[21] summarized some potential urban runoff effects reported in other studies. Two studies of a reservoir near Knoxville, Tennessee, showed that the quality of the contributing streams were degraded to a small extent by urban runoff and that the reservoir itself experienced a significant change in DO, pH, BOD_5, conductivity, temperature, total solids, and total coliform bacteria during short storm events. In another study at the Christina River in Newark, Delaware, cadmium and lead concentrations several miles below the urban area remained at elevated values up to 48 hours after storm periods. The quality of runoff from similar non-urbanized watersheds was compared with this urbanized area's runoff. They found that concentrations of nitrates, phosphorus, heavy metals and pesticides were considerably higher in the urbanized areas than in the forested regions. Field and Cibik[21] also reported on a study conducted in Virginia, where water, sediment, detritus, caddisflies, snails and crayfish were analyzed for iron, manganese, nickel, lead, cadmium, zinc, chromium and copper. The sampling areas were exposed to wastewater effluent and urban runoff. The found that concentrations increased immediately below stormwater discharge locations. They also reported on a study from Hawaii which indicated that receiving water conditions were designated as hazardous because of very high concentrations of suspended solids, heavy metals and bacterial pathogens.

During the Coyote Creek, San Jose study, dry weather concentrations of many constituents exceeded expected wet weather concentrations by factors of two to five times[22]. During dry weather, many of the major constituents (e.g., major ions, total solids, etc.) were significantly greater in both the urban and nonurban reaches. These constituents were all found in substantially lower concentrations in the urban runoff and in the rain. The rain and the resultant runoff apparently diluted the concentrations of these constituents in the creek during wet weather. Within the urban area, many constituents were found in greater concentrations during wet weather than during dry weather (Chemical Oxygen Demand, organic nitrogen, and especially heavy metals - lead, zinc, copper, cadmium, mercury, iron, and nickel). Lead concentrations were found to be more than seven times as great in the urban reach than in the nonurban reach during dry weather, with a confidence level of 75 percent. Other significant increases in urban area concentrations occurred for nitrogen, chloride, orthophosphate, Chemical Oxygen Demand, specific conductance, sulfate, and zinc. The dissolved oxygen measurements were about 20 percent less in the urban reach than in the nonurban reach of the creek.

Bolstad and Swank[23] examined the in-stream water quality at 5 sampling stations in Cowetta Creek in western North Carolina over a 3 year period. The watershed is 4350 ha and is relatively undeveloped (forested) in the area above the most upstream sampling station and becomes more urbanized at the downstream sampling station. Baseflow water quality was good, while most constituents increased during wet weather. Bacteria values increased substantially during wet weather, with total and fecal coliforms, and fecal streptococci increasing by two to three times during storms. Water quality was compared to building density for the different monitoring stations, with increasing stormwater contaminant concentrations (especially for turbidity, bacteria, and some inorganic solutes) with increasing building densities. Baseflow concentrations also typically increased with density, but at a much lower rate. In addition, the highest concentrations observed during individual events corresponded to the highest flow rates.

Reported Fish Kill Information

Urban runoff impacts are sometimes difficult for many people to appreciate in urban areas. Fish kills are the most obvious indication of water quality problems for many people. However, because urban receiving water quality is usually so poor, the aquatic life in typical urban receiving waters is usually limited in abundance and diversity, and quite resistant to poor water quality. Sensitive native organisms have typically been displaced, or killed, long ago. It is also quite difficult to identify the specific cause of a fish kill in an urban stream. Ray and White[24], for example, stated that one of the complicating factors in determining fish kills related to heavy metals is that the fish mortality may lag behind the first toxic exposure by several days, and is usually detected many miles downstream from the discharge location. The actual concentrations of the water quality constituents that may have caused the kill could then be diluted beyond detection limits, making probable sources of the toxic materials impossible to determine in many cases.

Heaney, *et al*.[18] reviewed fish kill information reported to government agencies during 1970 to 1979. They found that less than three percent of the reported 10,000 fish kills were identified as having been caused by urban runoff. This is less than 30 fish kills per year nationwide. A substantial number of these 10,000 fish kills were not identified as having any direct cause. They concluded that many of these fish kills were likely caused by urban runoff, or a combination of problems that could have been worsened by urban runoff.

During the Bellevue, Washington, receiving water studies, some fish kills were noted in the unusually clean urban streams[25]. The fish kills were usually associated with inappropriate discharges to the storm drainage system (such as cleaning materials and industrial chemical spills) and not from "typical" urban runoff. However, as noted later, the composition of the fish in the urban stream was quite different, as compared to the control stream[26].

Fish kill data have therefore not been found to be a good indication of receiving water problems caused by urban runoff. However, as discussed previously, the composition of the fisheries and other aquatic life taxonomic indicators are sensitive indicators of receiving water problems in urban streams.

Toxicological Effects of Stormwater

Even though acute toxicity of stormwater on most aquatic organisms has been relatively rare, short-term toxicity tests are still commonly conducted as part of some whole effluent toxicity (WET) tests required by some state regulatory agencies and by some stormwater researchers[147].

The need for endpoints for toxicological assessments using multiple stressors was discussed by Marcy and Gerritsen[27]. They used five watershed-level ecological risk assessments to develop appropriate endpoints based on specific project objectives. Dyer and White[28a] also examined the problem of multiple stressors affecting toxicity assessments. They felt that field surveys rarely can be used to verify simple single parameter laboratory experiments. They developed a watershed approach integrating numerous databases in conjunction with *in-situ* biological observations to help examine the effects of many possible causative factors. Toxic effect endpoints are additive for compounds having the same "mode of toxic action", enabling predictions of complex chemical mixtures in water, as reported by *Environmental Science & Technology*[28b]. They reported that EPA researchers at the Environmental Research Laboratory in Duluth, MN, identified about five or six major action groups that contain almost all of the compounds of interest in the aquatic environment. Much work still needs to be done, but these new analytical methods may enable the in-stream toxic effects of stormwater to be better predicted.

Ireland, *et al*.[29] found that exposure to UV radiation (natural sunlight) increased the toxicity of PAH contaminated urban sediments to *C. dubia*. The toxicity was removed when the UV wavelengths did not penetrate the water column to the exposed organisms. Toxicity was also reduced significantly in the presence of UV when the organic fraction of the stormwater was removed. Photo-induced toxicity occurred frequently during low flow conditions and wet weather, but was reduced during turbid conditions.

Johnson, *et al*.[30] and Herricks, *et al*.[10, 11] describe a structured tier testing protocol to assess both short-term and long-term wet weather discharge toxicity that they developed and tested. The protocol recognizes that the test systems must be appropriate to the time-scale of exposure during the discharge. Therefore, three time-scale protocols were developed, for intra-event, event, and long-term exposures. The use of standard whole effluent toxicity (WET) tests were found to over-estimate the potential toxicity of stormwater discharges.

The effects of stormwater on Lincoln Creek, near Milwaukee, WI, were described by Crunkilton, *et al*.[12]. Lincoln Creek drains a heavily urbanized watershed of 19 mi^2 that is about nine miles long. On-site toxicity testing was conducted with side-stream flow-through aquaria using fathead minnows, plus in-stream biological assessments, along with water and sediment chemical measurements. In the basic tests, Lincoln Creek water was continuously pumped through the test tanks, reflecting the natural changes in water quality during both dry and wet weather conditions. The continuous flow-through mortality tests indicated no toxicity until after about 14 days of exposure, with more than 80% mortality after about 25 days, indicating that the shorter-term toxicity tests likely underestimate stormwater toxicity. The biological and physical habitat assessments also supported a definitive relationship between degraded stream ecology and urban runoff.

Rainbow[31] presented a detailed overview of heavy metals in aquatic invertebrates. He concluded that the presence of a metal in an organism cannot tell us directly whether that metal is poisoning the organism. However, if compared to concentrations in a suite of well-researched biomonitors, it is possible to determine if the accumulated concentrations are atypically high, with a possibility that toxic effects may be present. Allen[32] also presented an overview of metal contaminated aquatic sediments. This book presents many topics that would enable the user to better interpret measured heavy metal concentrations in urban stream sediments.

One of the key objectives of the Chesapeake Bay restoration effort is to reduce the impacts of toxicants, for which stormwater is a recognized major source for the area. Hall, *et al*.[33] describe the *Toxics Reduction Strategy*, based on water column and sediment chemical analyses, benthic community health, and fish body burdens. More than 40% of the sites have displayed some degree of water column toxicity, and about 70% of the sites have displayed sediment toxicity. Garries, *et al*.[34] further describe how the list of *Toxics of Concern* is developed for Chesapeake Bay.

Sediment contaminated by stormwater discharges has a detrimental effect on the receiving water biological community. Schueler[35] summarized *in-situ* assessment methods of stormwater-impacted sediments. The use of *in-situ* test chambers, using *C. dubia*, eliminates many of the sample disruption problems associated with conducting sediment toxicity tests in the laboratory. Love and Woolley[36] found that stormwater was alarmingly more toxic than treated sewage and that treatment before reuse of residential area stormwater may be needed.

Pitt[37] reported a series of laboratory toxicity tests using 20 stormwater and CSO samples. He found that the most promising results are associated with using several complementary tests, instead of any one test method. However, simple screening toxicity tests (such as using the Azur Microtox test) are useful during preliminary assessments or for treatability tests.

Huber and Quigley[38] studied highway construction and repair materials (e.g. deck sealers, wood preservatives, waste-amended pavement, etc.) for their chemical and toxicological properties and leaching characteristics. *Daphnia magna* (a water flea) and the algae *Selenastrum capricornutum* were used for the toxicity tests. Leaching was evaluated as a function of time using batch tests, flat plate tests and column tests, as appropriate for the end-use of the highway material. These comprehensive tests identified a number of maintenance and construction materials that should be avoided for use near aquatic environments due to their elevated toxicity.

Kosmala, *et al*.[40] used *C. dubia* in laboratory toxicity tests in combination with field analysis of the *Hydropsychid* life cycle to assess the impact of both the wastewater treatment plant effluent and the stormwater overflow on the receiving water. They found that the results seen in the laboratory toxicity tests and in the *in-situ* biological measurements were due to nutrient and micropollutant loadings. Marsalek, *et al*.[41] used several different toxicity tests to assess the various types of toxicity in typical urban runoff and in runoff from a multi-lane highway. The tests included traditional toxicity analysis using *Daphnia magna*, the Microtox® toxicity test, sub-mitochondrial particles, and the SOS Chromotest for genotoxicity. Marsalek and Rochfort[42] also investigated the toxicity of urban stormwater and CSO. Acute toxicity, chronic toxicity and genotoxicity of stormwater and CSO were studied at 19 urban sampling sites in Ontario, Canada, using a battery of seven bioassays. Most frequent responses of severe toxicity were found in stormwater samples (in 14% of all samples), particularly those collected on freeways during the winter months. Compared to stormwater, CSO displayed lower acute toxicity (7% of the samples were moderately toxic, and none of the samples was severely toxic).

Skinner, *et al*.[43] showed that stormwater runoff produced significant toxicity in the early life stages of medaka (*Oryzias latipes*) and inland silverside (*Menidia beryllina*). Developmental problems and toxicity were strongly correlated with the total metal content of the runoff and corresponded with exceedences of water quality criteria of Cd, Cu, W, and Zn.

Tucker and Burton[44] compared *in-situ* versus laboratory conditions for toxicity testing of nonpoint-source runoff. They found that NPS runoff from urban areas was more toxic to the organisms in the laboratory while the agricultural runoff was more toxic to the organisms exposed *in-situ*. The differences seen between the two types of toxicity tests demonstrated the importance of *in-situ* assays in assessing the effects of NPS runoff. Hatch and Burton[45], using field and laboratory bioassays, demonstrated the impact of the urban stormwater runoff on *Hyalella azteca*, *Daphnia magna*, and *Pimephales promelas* survival after 48 hours of exposure. The significant toxicity seen at the outfall site was attributed to the contaminant accumulation in the sediments and the mobilization of the top layers of sediment during storm events.

Bickford, *et al*.[46] described the methodology developed and implemented by Sydney Water in Australia to assess the risk to humans and aquatic organisms in creeks, rivers, estuaries and ocean waters from wet weather flows (WWFs). The model used in this study was designed to predict concentrations of various chemicals in WWFs and compare the values to toxicity reference values. Brent and Herricks[47] proposed a methodology for predicting and quantifying the toxic response of aquatic systems to brief exposures to pollutants such as the contaminants contained in stormwater runoff. The method contains an event-focused toxicity method, a test metric (ETU, event toxicity unit) to represent the toxicity of intermittent events, and an event-based index that would described the acute toxicity of this brief exposure. The toxicity metric proposed (PE-LET50 [post-exposure lethal exposure time]) was the exposure duration required to kill 50% of the population during a pre-specified, post-exposure monitoring period. Colford, *et al*.[48] proposed three methods of analytically evaluating the impact of storm sewer and combined sewer outflows on public health, especially in areas that may receive through deposition the harmful agents in sewage and combined sewage.

Subtle (Chronic) Effects of Stormwater Discharges on Aquatic Life

Many studies have shown the severe detrimental effects of urban runoff on receiving water organisms. These studies have generally examined receiving water conditions above and below a city, or by comparing two parallel streams, one urbanized and another nonurbanized. The researchers usually carefully selected the urbanized streams to minimize contaminant sources other than urban runoff. However, few studies have examined direct cause and effect relationships of urban runoff for receiving water aquatic organisms[49]. The following paragraphs briefly describe a variety of urban receiving water investigations.

Klein[50] studied 27 small watersheds having similar physical characteristics, but having varying land uses, in the Piedmont region of Maryland. During an initial phase of the study, they found definite relationships between water quality and land use. Subsequent study phases examined aquatic life relationships in the watersheds. The principal finding was that stream aquatic life problems were first identified with watersheds having imperviousness areas comprising at least 12 percent of the watershed. Severe problems were noted after the imperviousness quantities reached 30 percent.

Receiving water impact studies were also conducted in North Carolina[51-53]. The benthic fauna occurred mainly on rocks. As sedimentation increased, the amount of exposed rocks decreased, with a decreasing density of benthic macroinvertebrates. Data from 1978 and 1979 in five cities showed that urban streams were grossly polluted by a combination of toxicants and sediment. Chemical analyses, without biological analyses, would have underestimated the severity of the problems because the water column quality varied rapidly, while the major problems were associated with sediment quality and effects on macroinvertebrates. Macroinvertebrate diversities were severely reduced in the urban streams, compared to the control streams. The biotic indices indicated very poor conditions for all urban streams. Occasionally, high populations of pollutant-tolerant organisms were found in the urban streams, but would abruptly disappear before subsequent sampling efforts. This was probably caused by intermittent discharges of spills or illegal dumpings of toxicants. Although the cities studied were located in different geographic areas of North Carolina, the results were remarkably uniform.

During the Coyote Creek, San Jose, California, receiving water study, 41 stations were sampled in both urban and nonurban perennial flow stretches of the creek over three years. Short and long-term sampling techniques were used to evaluate the effects of urban runoff on water quality, sediment properties, fish, macroinvertebrates, attached algae, and rooted aquatic vegetation[22]. These investigations found distinct differences in the taxonomic composition and relative abundance of the aquatic biota present. The non-urban sections of the creek supported a comparatively diverse assemblage of aquatic organisms including an abundance of native fishes and numerous benthic macroinvertebrate taxa. In contrast, however, the urban portions of the creek (less than 5% urbanized), affected only by urban runoff discharges and not industrial or municipal discharges, had an aquatic community generally lacking in diversity and was dominated by pollution-tolerant organisms such as mosquitofish and tubificid worms.

A major nonpoint runoff receiving water impact research program was conducted in Georgia[54]. Several groups of researchers examined streams in major areas of the state. Benke, *et al*.[55] studied 21 stream ecosystems near Atlanta having watersheds of one to three square miles each and land uses ranging from 0 to 98 percent urbanization. They measured stream water quality but found little relationship between water quality and degree of urbanization. The water quality parameters also did not identify a major degree of pollution. In contrast, there were major correlations between urbanization and the number of species found. They had problems applying diversity indices to their study because the individual organisms varied greatly in size (biomass). CTA[56] also examined receiving water aquatic biota impacts associated with urban runoff sources in Georgia. They studied habitat composition, water quality, macroinvertebrates, periphyton, fish, and toxicant concentrations in the water, sediment, and fish. They found that the impacts of land use were the greatest in the urban basins. Beneficial uses were impaired or denied in all three urban basins studied. Fish were absent in two of the basins and severely restricted in the third. The native macroinvertebrates were replaced with pollution tolerant organisms. The periphyton in the urban streams were very different from those found in the control streams and were dominated by species known to create taste and odor problems.

Pratt, *et al*.[57] used basket artificial substrates to compare benthic population trends along urban and nonurban areas of the Green River in Massachusetts. The benthic community became increasing disrupted as urbanization increased. The problems were not only associated with times of heavy rain, but seemed to be affected at all times. The stress was greatest during summer low flow periods and was probably localized near the stream bed. They concluded that the high degree of correspondence between the known sources of urban runoff and the observed effects on the benthic community was a forceful argument that urban runoff was the causal agent of the disruption observed.

Cedar swamps in the New Jersey Pine Barrens were studied by Ehrenfeld and Schneider[58]. They examined nineteen wetlands subjected to varying amounts of urbanization. Typical plant species were lost and replaced by weeds and exotic plants in urban runoff affected wetlands. Increased uptakes of phosphorus and lead in the plants were found. It was concluded that the presence of stormwater runoff to the cedar swamps caused marked changes in community structure, vegetation dynamics, and plant tissue element concentrations.

Medeiros and Coler[59] and Medeiros, *et al*.[60] used a combination of laboratory and field studies to investigate the effects of urban runoff on fathead minnows. Hatchability, survival, and growth were assessed in the laboratory in flow-through and static bioassay tests. Growth was reduced to one half of the control growth rates at 60 percent dilutions of urban runoff. The observed effects were believed to be associated with a combination of toxicants.

The University of Washington[25,61-67] conducted a series of studies to contrast the biological and chemical conditions in urban Kelsey Creek with rural Bear Creek in Bellevue, Washington. The urban creek was significantly degraded when compared to the rural creek, but still supported a productive, but limited and unhealthy salmonid fishery. Many of the fish in the urban creek, however, had respiratory anomalies. The urban creek was not grossly polluted, but flooding from urban developments had increased dramatically in recent years. These increased flows dramatically changed the urban stream's channel, by causing unstable conditions with increased stream bed movement, and by altering the availability of food for the aquatic organisms. The aquatic organisms were very dependent on the few relatively undisturbed reaches. Dissolved oxygen concentrations in the sediments depressed embryo salmon survival in the urban creek. Various organic and metallic priority pollutants were discharged to the urban creek, but most of them were apparently carried through the creek system by the high storm flows to Lake Washington. The urbanized Kelsey Creek also had higher water temperatures (probably due to reduced shading) than Bear Creek. This probably caused the faster fish growth in Kelsey Creek.

The fish population in the urbanized Kelsey Creek had adapted to its degrading environment by shifting the species composition from Coho salmon to less sensitive cutthroat trout and by making extensive use of less disturbed refuge areas. Studies of damaged gills found that up to three-fourths of the fish in Kelsey Creek were affected with respiratory anomalies, while no cutthroat trout and only two of the Coho salmon sampled in the forested Bear Creek had damaged gills. Massive fish kills in Kelsey Creek and its tributaries were also observed on several occasions during the project due to the dumping of toxic materials down the storm drains.

There were also significant differences in the numbers and types of benthic organisms found in urban and forested creeks during the Bellevue research. Mayflies, stoneflies, caddisflies, and beetles were rarely observed in the urban Kelsey Creek, but were quite abundant in the forested Bear Creek. These organisms are commonly regarded as sensitive indicators of environmental degradation. One example of degraded conditions in Kelsey Creek was shown by a species of clams (*Unionidae*) that was not found in Kelsey Creek, but was commonly found in Bear Creek. These clams are very sensitive to heavy siltation and unstable sediments. Empty clam shells, however, were found buried in the Kelsey Creek sediments indicating their previous presence in the creek and their inability to adjust to the changing conditions. The benthic organism composition in Kelsey Creek varied radically with time and place while the organisms were much more stable in Bear Creek.

Urban runoff impact studies were conducted in the Hillsborough River near Tampa Bay, Florida, as part of the U.S. EPA's Nationwide Urban Runoff Program (NURP)[68]. Plants, animals, sediment, and water quality were all studied in the field and supplemented by laboratory bioassay tests. Effects of salt water intrusion and urban runoff were both measured because of the estuarine environment. During wet weather, freshwater species were found closer to the Bay than during dry weather. In coastal areas, these additional natural factors made it even more difficult to identify the cause and effect relationships for aquatic life problems. During another NURP project, Striegl[69] found that the effects of accumulated contaminants in Lake Ellyn (Glen Ellyn, Ill.) inhibited desirable benthic invertebrates and fish and increased undesirable phyotoplankton blooms.

The number of benthic organism taxa in Shabakunk Creek in Mercer County, New Jersey, declined from 13 in relatively undeveloped areas to four below heavily urbanized areas[70,71]. Periphyton samples were also analyzed for heavy metals with significantly higher metal concentrations found below the heavily urbanized area than above.

Stewart, *et al*.[72] collected diatoms (*Bacillariophyta*) and water quality samples from three streams that drain the Great Marsh in the Indiana Dunes National Lakeshore. They found that diatom species diversity could be used as indicators of water quality, which could then be linked to land use in a watershed. Diatom species diversity was most variable in areas with poorer water quality and was directly correlated to the total alkalinity, total hardness and specific conductance of the water in the stream.

A number of papers presented at the 7th International Conference on Urban Storm Drainage, held in Hannover, Germany, described receiving water studies that investigated organic and heavy metal toxicants. Handová, *et al*.[73] examined the bioavailability of metals from CSOs near Prague. They compared these results with biomonitoring. The metals were ranked according to their mobility as: Cd (95%), Zn (87%), Ni (64%), Cr (59%), Pb (48%), and Cu (45%). The mobile fraction was defined as the metal content that was exchangeable, bound to carbonates, bound to iron and manganese oxides, and bound to organic matter. Boudries, *et al*.[74] and Estèbe, *et al*.[75] investigated heavy metals and organics bound to particulates in the River Seine near Paris. The Paris CSOs caused a significant increase in the aliphatic and aromatic hydrocarbons bound to river sediments. The high flows during the winter were associated with lower heavy metal associations with the sediment, compared to the lower summer flow conditions. These differences were found to be due to dilution of the CSOs in the river and to the changing contributions of rural versus urban suspended solids during the different seasons.

The Northeastern Illinois Planning Commission[76] compared comprehensive fish survey information from over 40 northeastern Illinois small to moderate-sized streams and rivers to demographic data for the contributing watershed areas. The streams had watershed areas ranging from about 12 to 222 square miles and had population densities ranging from about 30 to more than 4,500 people per square mile. The fish data was used in the index of biotic integrity (IBI) to identify the quality of the fish populations. Table 1 lists the fish data that is used in the IBI and Table 2 shows the different scores for the quality categories. Factors necessary for good and excellent quality fish communities include the presence of diverse and reproducing fish and other aquatic organisms, including a significant percentage of intolerant species (such as darters and smallmouth bass).

Table 1. Index of Biotic Integrity (IBI) Metrics[76]

Category	Metric
Species richness and composition	Total number of fish species
	Number and identity of darter species
	Number and identity of sunfish species
	Number and identity of sucker species
	Number and identity of intolerant species
	Proportion of individuals as green sunfish
Trophic composition	Proportion of individuals as omnivores
	Proportion of individuals as hybrids
	Proportion of individuals as piscivores
Fish abundance and condition	Number of individuals in sample
	Proportion of individuals as hybrids
	Proportion of individuals with disease, tumors, fin damage, and skeletal anomalies

Table 2. Illinois Environmental Protection Agency (IEPA) Biological Stream Characterization (BSC) and Index of Biotic Integrity (IBI) Classifications and Criteria[76]

IBI Score	Stream Class	BSC Category	Biotic Resource Quality
51 – 60	A	Unique	Excellent
41 – 50	B	Highly Valued	Good
31 – 40	C	Moderate	Fair
21 – 30	D	Limited	Poor
≤ 20	E	Restricted	Very Poor

The more commonly used imperviousness-based indicator of development was not used due to a lack of available data and the difficulty of acquiring good quality current imperviousness data, let alone estimating historical imperviousness data. In contrast, population data was readily available and thought to be an adequate indicator of the extent and density of urbanization in the watersheds. They found that nearly all streams in urban and suburban watersheds having population densities greater than about 300 people per square mile showed signs of considerable impairment to their fish communities (being in fair to very poor condition). In contrast, nearly all rural streams supported fish communities that were rated good or excellent. They identified both point and nonpoint sources as major contributors to these impairments. However, the point source discharges and CSO discharges have substantially decreased over the past 20 years, while the nonpoint source discharges have increased significantly with increased development, and the fisheries are still declining in many areas. In stable areas that were mostly affected by point sources and CSOs, documented dramatic improvements in some water quality indicators (especially DO and ammonia), and the fish populations, have occurred. In similar areas, but having continued urban development, the fisheries have continued to decline.

They concluded that although rural watersheds have known water quality problems (especially agricultural chemicals and erosion, plus manure runoff), these issues did not prevent the attainment of mostly high quality fisheries in these areas. Similar conclusions were noted in the comparison study by the USGS in North Carolina[77] of forested, agricultural, and urban streams. Although the forested streams were of the best quality, the streams in the agricultural areas were of intermediate quality and had significantly better biological conditions than the urban stream (which had poor macroinvertebrate and fish conditions, poor sediment and temperature conditions, and fair substrate and nutrient conditions).

Habitat Effects Caused by Stormwater Discharges

Some of the most serious effects of urban runoff are on the aquatic habitat of the receiving waters. These habitat effects are in addition to the pollutant concentration effects. Numerous papers already referenced found significant sedimentation problems in urban receiving waters. The major effects of urban sediment on the aquatic habitat include silting of spawning and food production areas and unstable bed conditions[78]. Other major habitat destruction problems include rapidly changing flows and the absence of refuge areas to protect the biota during these flow changes. Removal of riparian vegetation can increase water temperatures and a major source of large organic debris that are important refuge areas. The major references on stream geomorphology that many of the following researchers based their work on were by Leopold, *et al*.[79], Brookes[80], and Rosgen[81]. These fundamental references should be consulted for excellent descriptions of the many natural processes affecting streams in transition. Brookes also specifically examines urbanization effects on stream morphology. Knowledge of these basic processes will better enable an understanding of local stream changes occurring with watershed urbanization. This understanding will, in turn, enable more efficient rehabilitation efforts of degraded streams and the use of watershed controls to minimize these effects.

Brookes[80] has documented many cases in the U.S. and Great Britain of stream morphological changes associated with urbanization. These changes are mostly responsible for habitat destruction that are usually the most significant detriment to aquatic life. In many cases, water quality improvement would result in very little aquatic life benefits if the physical habitat is grossly modified. The most obvious habitat problems are associated with stream "improvement" projects, ranging from removal of debris, to straightening streams, to channelization projects. Brookes[80,82] presents a number of ways to minimize habitat problems associated with stream channel projects, including stream restoration.

Wolman and Schick[83] observed deposition of channel bars, erosion of channel banks, obstruction of flows, increased flooding, shifting of channel bottoms, along with concurrent changes in the aquatic life, in Maryland streams affected by urban construction activities.

Pess and Bilby[84] identified Coho salmon (*Oncorhynchus kisutch*) distribution and abundance in Puget Sound rivers and explained the distribution by using both stream-reach and watershed-scale habitat characteristics, including the influence of urban areas on the habitat. In the Puget Sound region of the U.S. Pacific Northwest, Greenberg, *et al*.[85] developed and evaluated the Urban Stream Baseline Evaluation Method to characterize baseline habitat conditions for salmonids. The methodology, based on assessment of geomorphic suitability, fish distribution and habitat alteration, was recommended for use to prioritize recovery actions.

Bragg and Kershner[86] investigated the impact on the habitats of aquatic life and they found that coarse woody debris in riparian zones can be used successfully to maintain the integrity of these ecosystems. Larson[87] evaluated the effectiveness in urban areas of these habitat restoration activities using large woody debris and found that in urban areas, the success of restoration may be hindered by the high sediment loads and increased flow associated with urbanization. Markowitz, *et al*.[88] documented the CSO Long Term Control Plan implemented by the City of Akron, Ohio which focused on habitat preservation and aquatic life use of the receiving waters. The plan included these non-traditional alternatives: riparian setbacks in undeveloped areas, stream restoration, linear parks or greenways and artificial riffles for stream aeration, and were found to cost less than five percent of the typical cost of controlling CSO flows. A methodology to investigate the chronic and cumulative degradation of the river Orne due to CSO and urban runoff was presented by Zobrist, *et al.*[89], with the results being used to evaluate management activities. Xu, *et al.*[91] reported on the improvement plan being used for a river passing through the downtown area of a city in Western Japan and the problems that were inherent with developing a compromise strategy between flood control and mitigation and the desire to have an attractive waterway through the city. The final improvement plan recommended construction of a new flood drain tunnel and a new underground flood control reservoir.

Cianfrani, *et al.*[92] used a GIS system to document the results of a comprehensive inventory of the natural resources of the Fairmount Park (Philadelphia, Pennsylvania) stream system, including vegetation communities, fish, aquatic and terrestrial insects, birds, mollusks, amphibians, reptiles, and streams. The stream assessment also included the characterization of stream reaches by in-stream habitat, geomorphology and riparian zone. This GIS inventory then was used in planning the restoration of sites in the Fairmount Park system. Derry, *et al.*[93] reported on the habitat management strategies implemented by the City of Olympia, Washington, to control the degradation of aquatic habitats by urban stormwater runoff. These management strategies provided a basis for resolving the conflict between growth and the protection of aquatic resources. Ishikawa, *et al.*[94] reported on the efforts to restore the hydrological cycle in the Izumi River Basin in Yokohama, Japan while Saeki, *et al.*[95] have documented the efforts of the Tokyo Metropolitan Government and its Basin Committee to restore the natural water cycle in the Kanda River. Kennen[96] investigated the relationship between selected basin and water-quality characteristics in New Jersey streams and the impact on the macroinvertebrate community and its habitat. He found that urban areas had the greatest probability of having impacted stream areas, with the amount of urban land and the total flow of treated sewage effluent being the strongest explanatory variables for the impact. He also found that levels of impairment were significantly different between the Atlantic Coastal Rivers drainage area and the Lower Delaware River drainage area.

Increased Flows from Urbanization

Increased flows are the probably the best know example of impacts associated with urbanization. Most of the recognition has of course focused on increased flooding and associated damages. This has led to numerous attempts to control peak flows from new urban areas through the use of regulations that limit post development peak flows to pre development levels for relatively large design storms. The typical response has been to use dry detention ponds. This approach is limited, and may actually increase downstream flows. In addition to the serious issue of flooding, high flows also cause detrimental ecological problems in receiving waters. The following discussion presents several case studies where increased flows were found to have serious effects on stream habitat conditions.

The aquatic organism differences in urbanized and control streams found during the Bellevue Urban Runoff Program were probably most associated with the increased peak flows. The increased flows in the urbanized Kelsey Creek resulted in increases in sediment carrying capacity and channel instability of the creek[61-65]. Kelsey Creek had much lower flows than the control Bear Creek during periods between storms. About 30 percent less water was available in Kelsey Creek during the summers. These low flows may also have significantly affected the aquatic habitat and the ability of the urban creek to flush toxic spills or other dry weather contaminants from the creek system[66, 67]. Kelsey Creek had extreme hydrologic responses to storm. Flooding substantially increased in Kelsey Creek during the period of urban development; the peak annual discharges almost doubled in the last 30 years, and the flooding frequency also increased due to urbanization[66, 67]. These increased flows in urbanized Kelsey Creek resulted in greatly increased sediment transport and channel instability.

Bhaduri, *et al.*[97] also quantified the changes in streamflow and associated decreases in groundwater recharge associated with urbanization. They point out that the most widely addressed hydrologic effect of urbanization is the peak discharge increases that cause local flooding. However, the increase in surface runoff volume also represents a net loss in groundwater recharge. They point out that urbanization is linked to increased variability in volume of water available for wetlands and small streams, causing "flashy" or "flood-and-drought" conditions. In northern Ohio, urbanization at a study area was found to cause a 195% increase in the annual volume of runoff, while the expected increase in the peak flow for the local 100-yr event was only 26% for the same site. Although any increase in severe flooding is problematic and cause for concern, the much larger increase in annual runoff volume, and associated decrease in groundwater recharge, likely has a much greater effect on in-stream biological conditions.

Snodgrass, *et al*.[16] reported that in the Toronto, Ontario, area, flows causing bankfull conditions occur with a return frequency of about 1.5 years. Storms with this frequency are in general equilibrium with resisting forces that tend to stabilize the channel (such as vegetation and tree root mats), with increased flows overcoming these resisting forces causing channel enlargement. Infrequent flows can therefore be highly erosive. With urbanization, the flows that were bankfull flows during historical times now occur much more frequently (about every 0.4 years in Toronto). The channel cross-sectional area therefore greatly increases to accommodate the increased stream discharges and power associated with the "new" 1.5 year flows that are trying to re-establish equilibrium.

Booth and Jackson[98] examined numerous data from lowland streams in western Washington and concluded that development having about 10% imperviousness caused a readily apparent degradation of aquatic life in the receiving waters. They linked the association between increased imperviousness and biological degradation to increases in flows and sediment discharges. They concluded that conventional methods to size stormwater mitigation measures (especially detention ponds) were seriously inadequate. They felt that without a better understanding of the critical processes that lead to degradation, some downstream damage to the aquatic ecosystem is likely inevitable, without unpopular restrictions to the extent of development in the watershed corresponding to <10% imperviousness. The stream channels were generally stable if the effective impervious areas remained below 10% of the complete watershed. This level of development corresponds to a 2-year developed condition flow being less than the historical 10-year pre-developed flow condition. They found that the classical goal of detention ponds to maintain predevelopment flows was seriously inadequate because there is no control on the duration of the peak flows. They showed that a duration standard to maintain post development flow durations for all sediment-transporting discharges to predevelopment durations will avoid many receiving water habitat problems associated with stream instability. Without infiltration, the amount of runoff will obviously still increase with urbanization, but the increased water could be discharged from detention facilities at flow rates below the critical threshold causing sediment transport. The identification of the threshold discharge below which sediment transport does not occur, unfortunately, if difficult and very site specific. A presumed threshold discharge of about one-half of the pre-development 2-year flow was recommended for gravel bedded streams. Sand-bedded channels have sediment transport thresholds that are very small, with inevitable bed load transport likely to occur for most levels of urbanization.

Channel Modifications due to Urban Wet Weather Flow Discharges

Changes in physical stream channel characteristics can have a significant effect on the biological health of the stream. These changes in urban streams have been mostly related to changes in the flow regime of the stream, specifically increases in peak flow rates, increased frequencies and durations of erosive flows, and channel modifications made in an attempt to accommodate increased stormwater discharges.

Schueler[99] stated that channel geometric stability can be a good indicator of the effectiveness of stormwater control practices. He also found that once a watershed area has more than about 10 to 15% effective impervious cover, noticeable changes in channel morphology occur, along with quantifiable impacts on water quality, and biological conditions. Stephenson[100] studied changes in streamflow volumes in South Africa during urbanization. He found increased stormwater runoff, decreases in the groundwater table, and dramatically decreased times of concentration. The peak flow rates increased by about two-fold, about half caused by increased pavement (in an area having only about 5% effective impervious cover), with the remainder caused by decreased times of concentration (related to the increased drainage efficiency of artificial conveyances).

Richey[64] made some observations about bank stabilities in Kelsey and Bear Creeks as part of the Bellevue, WA, NURP project[25]. She notes that the Kelsey Creek channel width had been constrained during urban development. Thirty-five percent of the urbanized Kelsey Creek channel mapped during these projects was modified by the addition of some type of stabilization structure. Only eight percent of non-urbanized Bear Creek's length was stabilized. Most of the stabilization structures in Bear Creek were low walls in disrepair while more than half of the structures observed along Kelsey Creek were large riprap or concrete retention walls. The necessity of the stabilization structures was evident from the extent and severity of erosion cuts and the number of deposition bars observed along the Kelsey Creek stream banks. Bridges and culverts were also frequently found along Kelsey

Creek; these structures further act to constrict the channel. As discharges increased and the channel width is constrained, the velocity increases, causing increases in erosion and sediment transport.

The use of heavy riprapping along the creek seemed to worsen the flood problems. Storm flows are unable to spread out onto the flood plain and the increased velocities are evident downstream along with increased sediment loads. This rapidly moving water has enough energy to erode unprotected banks downstream of riprap. Many erosion cuts along Kelsey Creek downstream of these riprap structures were found. Similar erosion of the banks did not occur in Bear Creek. Much of the Bear Creek channel had a wide flood plain with many side sloughs and back eddies. High flows in Bear Creek could spread onto the flood plains and drop much of their sediment load as the water velocities decreased.

The University of Washington studies also examined sediment transport in urbanized Kelsey and non-urbanized Bear Creeks. Richey[64] found that the relative lack of debris dams and off-channel storage areas and sloughs in Kelsey Creek contributed to the rapid downstream transit of water and materials. The small size of the riparian vegetation and the increased stream power probably both contributed to the lack of debris in the channel. It is also possible that the channel debris may have been cleared from the stream to facilitate rapid drainage. The high flows from high velocities caused the sediments to be relatively coarse. The finer materials were more easily transported downstream. Larger boulders were also found in the sediment but were probably from failed riprap or gabion structures.

Maxted[101] examined stream problems in Delaware associated with urbanization. He found an apparent strong correlation between habitat score and biology score from 40 stream study locations. He found that it is not possible to have acceptable biological conditions if the habitat is degraded. The leading contributor to habitat degradation was found to be urban runoff, especially the associated high flows and sediment accumulations.

A number of presentations concerning aquatic habitat effects from urbanization were made at the *Effects of Watershed Development and Management on Aquatic Ecosystems* conference held in Snowbird, UT, in August of 1996, sponsored by the Engineering Foundation and the ASCE. MacRae[102] presented a review of the development of the common zero runoff increase (ZRI) discharge criterion, referring to peak discharges before and after development. This criterion is commonly met using detention ponds for the 2 yr storm. MacRae shows how this criterion has not effectively protected the receiving water habitat. He found that stream bed and bank erosion is controlled by the frequency and duration of the mid-depth flows (generally occurring more often than once a year), not the bank-full condition (approximated by the 2 yr event). During monitoring near Toronto, he found that the duration of the geomorphically significant pre-development mid-bankfull flows increased by a factor of 4.2 times, after 34% of the basin had been urbanized, compared to before development flow conditions. The channel had responded by increasing in cross-sectional area by as much as 3 times in some areas, and was still expanding. Table 3 shows the modeled durations of critical discharges for predevelopment conditions, compared to current and ultimate levels of development with “zero runoff increase” controls in place. At full development and even with full ZRI compliance in this watershed, the hours exceeding the critical mid-bankfull conditions will increase by a factor of 10, with resulting significant effects on channel stability and the physical habitat.

Table 3. Hours of Exceedence of Developed Conditions with Zero Runoff Increase Controls Compared to Predevelopment Conditions[102]

Recurrence Interval (yrs)	Existing Flowrate (m3/s)	Exceedence for Predevelopment Conditions (hrs per 5 yrs)	Exceedence for Existing Development Conditions, with ZRI Controls (hrs per 5 yrs)	Exceedence for Ultimate Development Conditions, with ZRI Controls (hrs per 5 yrs)
1.01 (critical mid-bankfull conditions)	1.24	90	380	900
1.5 (bankfull conditions)	2.1	30	34	120

MacRae[102] also reported other studies that found that channel cross-sectional areas began to enlarge after about 20 to 25% of the watershed was developed, corresponding to about a 5% impervious cover in the watershed. When the watersheds are completely developed, the channel enlargements were about 5 to 7 times the original cross-sectional areas. Changes from stable streambed conditions to unstable conditions appear to occur with basin imperviousness of about 10%, similar to the value reported previously for serious biological degradation. He also summarized a study conducted in British Columbia that examined 30 stream reaches in natural areas, in urbanized areas having peak flow attenuation ponds, and in urbanized areas not having any stormwater controls. The channel widths in the uncontrolled urban streams were about 1.7 times the widths of the natural streams. The streams having the ponds also showed widening, but at a reduced amount compared to the uncontrolled urban streams. He concluded that an effective criterion to protect stream stability (a major component of habitat protection) must address mid-bankfull events, especially by requiring similar durations and frequencies of stream power (the product of shear stress and flow velocity, not just flow velocity alone) at these depths, compared to satisfactory reference conditions.

Much research on habitat changes and rehabilitation attempts in urban streams has occurred in the Seattle area of western Washington over the past 20 years. Sovern and Washington[103] described the in-stream processes associated with urbanization in this area, as part of a paper describing a recommended approach for the rehabilitation of urban streams. They were concerned that many "restoration" attempts of urban streams were destined to failure because of a lack of understanding of the actual changes occurring in streams as the watersheds changed from forested to urban land uses. They presented a concept of the "new urban stream" that attempts to correct several of the most important changes to better accommodate the native Pacific Northwest fish, instead of the unrealistic goal of trying to totally restore the steams to predevelopment conditions. The important factors that affect the direction and magnitude of the changes in a steam's physical characteristics due to urbanization include:

- the depths and widths of the dominant discharge channel will increase directly proportional to the water discharge. The width is also directly proportional to the sediment discharge. The channel width divided by the depth (the channel shape) is also directly related to sediment discharge.
- the channel gradient is inversely proportional to the water discharge rate, and is directly proportional to the sediment discharge rate and the sediment grain size.
- the sinuosity of the stream is directly proportional to the stream's valley gradient and is inversely proportional to the sediment discharge.
- bed load transport is directly related to the stream power and the concentration of fine material, and inversely proportional to the fall diameter of the bed material.

In their natural state, small streams in forested watersheds in Western Washington have small low-flow channels (the aquatic habitat channel) with little meandering[103]. The stream banks are nearly vertical because of clayey bank soils and heavy root structures, and the streams have numerous debris jams from fallen timber. The widths are also narrow, generally from 3 to 6 feet wide. Stable forested watersheds also support about 250 aquatic plant and animal species along the stream corridor. Pool/riffle habitat is dominant along streams having gradients less than about 2 percent slope, while pool/drop habitat is dominant along streams having gradients from 4 to 10 percent. The pools form behind large organic debris (LOD) or rocks. The salmon and trout in Western Washington have evolved to take advantage of these stream characteristics. Sovern and Washington[103] point out that less athletic fish species (such as chum and pink salmon) cannot utilize the steeper gradient, upper reaches, of the streams. Coho, steelhead and cutthroat can use these upper areas, however.

Urbanization radically affects many of these natural stream characteristics. Pitt and Bissonnette[25] reported that the Coho salmon and cutthroat trout were affected by the increased nutrients and elevated temperatures of the urbanized streams in Bellevue, as studied by the University of Washington as part of the U.S. EPA's NURP project[104]. These conditions were probably responsible for accelerated growth of the fry which were observed to migrate to Puget Sound and the Pacific Ocean sooner than their counterparts in the control forested watershed that was also studied. However, the degradation of sediments, mainly the decreased particle sizes, adversely affected their spawning areas in streams that had become urbanized. Sovern and Washington[103] reported that, in Western Washington, frequent high flow rates can be 10 to 100 times the predevelopment flows in urbanized areas, but that the low flows in the urban streams are commonly lower than the predevelopment low flows. They have concluded that the effects of urbanization on western Washington streams are dramatic, in most cases permanently changing the stream hydrologic balance by: increasing the annual water volume in the stream, increasing the volume and rate of storm flows, decreasing the low flows during dry periods, and increasing the sediment and contaminant discharges from the watershed. With urbanization, the streams increase in cross-sectional area to accommodate these increased flows and headwater downcutting occurs to decrease the channel gradient. The gradients of stable urban streams are often only about 1 to 2 percent, compared to 2 to 10 percent gradients in natural areas. These changes in width and the downcutting result in very different and changing stream conditions. The common pool/drop habitats are generally replaced by pool/riffle habitats, and the stream bed material is comprised of much finer material, for example. Along urban streams, fewer than 50 aquatic plant and animal species are usually found. They have concluded that once urbanization begins, the effects on stream shape are not completely reversible. Developing and maintaining quality aquatic life habitat, however, is possible under urban conditions, but it requires human intervention and it will not be the same as for forested watersheds.

Other Seattle area researchers have specifically examined the role that large woody debris (LWD) has in stabilizing the habitat in urban streams. Booth, *et al.*[105] found that LWD performs key functions in undisturbed streams that drain lowland forested watersheds in western Washington. These important functions include: energy dissipation of the flow energy, channel bank and bed stabilization, sediment trapping, and pool formation. Urbanization typically results in the almost complete removal of this material. They point out that logs and other debris have long been removed from channels in urban areas for many reasons, especially because of their potential for blocking culverts or to form jams at bridges, they may increase bank scour, and many residents favor "neat" stream bank areas (a lack of woody debris in and near the water and even with mowed grass to the waters edge). Booth, *et al*.[105] present and modify the stream classification system originally developed by Montgomery and Buffington[106] that recognizes LWD as an important component of Pacific northwest streams that are being severely affected by urbanization.

The role of LWD varies in each channel type, and the effects of its removal also varies. The channel types are described as follows. The upper colluvial channels are wholly surrounded by colluvium (sediment transported by creep or landsliding, and not by stream transport) and generally lie at the top of the channel network. The cascade channels are the steepest of the alluvial channels and are characterized as having tumbling flows around individual boulders that dissipate most of the energy of the flowing water. Only very small pools are in cascade channels. The step-pool channels have accumulations of debris that form a series of steps that are one to four channel widths apart. The steps separate small pools that accumulate fine sediment. The fine sediment can be periodically flushed downstream during rare events. "Free" step-pool channels are characterized by steps that are made of alluvium that can be periodically transported downstream during high flows, while "forced" step-pool channels are characterized by steps that are made of immovable obstructions (large logs or bedrock). The removal of LWD from a forced step-pool stream in the Cascade Range could be naturally compensated by the common occurrence of large boulders that also form forced steps. However, in the lowlands near Puget Sound, the available sand and gravel stream deposits are too small to form stable steps, and the removal of LWD would have a much more severe effect on the channel stability. Plane-bed channels have long and channel-wide reaches of uniform riffles and do not have pronounced meanders and associated pools. Pool-riffle channels are the most common lowland stream channels in western Washington. These streams have pronounced meanders with pools at the outside of the bends and corresponding bars on the inside of the bends. Riffles form in the relatively straight stretch between the pools. There are also "free" and "forced" pool-riffle channels. Forced riffle-pool channels are typically formed with obstructions, such as LWD, and their removal would generally lead to a plane-bed channel characteristic. Forced riffle-pool channels form due to natural meanders and the inertial forces of the water. Dune-ripple channels have beds mostly made of sand where the character of the bed material changes in response to the flows.

The role of LWD is also highly dependent on the width of the stream. In narrow channels (high gradient colluvial and cascade channels), much of the LWD can be suspended above the flows, rarely being submerged and not available as a fish refuge, a sediment trap, or to dissipate the water's energy. In wide channels (dune-ripple channels), the LWD may be significantly shorter than the channel width, with minimal stable opportunities to provide steps in the channel. Therefore, LWD plays a much more important role in channels having medium widths (lowland streams having plane-bed and pool-riffle channels), where the timber can become tightly lodged in the common flow channel. The removal of the LWD in these streams, especially in streams having few boulder steps, would have significant effects. Fish populations decline rapidly and precipitously following the removal of LWD in these critical streams [105].

Horner, *et al*.[107] described an extensive study in the Pacific Northwest where 31 stream reaches were examined since 1994 for a variety of in-stream and watershed characteristics. They felt that the most severe in-stream biological changes were most likely associated with changes in habitat, especially increased frequencies and magnitudes of high flows. These flow changes were therefore thought to most related to watershed factors affecting runoff, especially the amount of impervious areas in the watershed. They felt that the most rapid changes in ecological conditions were most likely to occur for urbanizing streams at relatively low levels of development, conditions representing most of the selected study sites.

Horner, *et al*.[107] found a rapid decline in biological conditions as total imperviousness area increases to about 8% in the watershed. The rate of decline is less for higher levels of urbanization. Eight study areas had better biological conditions than expected and were associated with higher amounts of intact wetlands along the riparian corridors than other sites, indicating a possible significant moderating effect associated with preserving stream corridors in their natural condition. The less tolerant Coho salmon is much more abundant than the more tolerant cutthroat trout only for very low levels of urbanization. Stormwater concentrations of zinc were also seen to increase steadily with increasing impervious areas. However, the concentrations are well below the critical water quality criteria until the impervious cover reaches about 40%, a level much greater than when significant biological effects are noted. Similar conclusions were made with other metal concentrations and contaminant concentrations in the sediment. They interpreted these findings to imply that contaminant conditions were much less important than habitat destruction when affecting in-stream biological conditions. They concluded that the preponderance of physical and biological evidence indicated rapid in-stream biological conditions at early stages of urbanization. However, chemical contaminants did not appear to significantly affect biological conditions in the early stages of urbanization, but may have at very high levels of urbanization. Based on their results, they developed a preliminary summary of the conditions that would allow high levels of biological functions in the Puget Sound area:

- total impervious areas less than 5% of the watershed area, unless mitigated by extensive riparian protection, management efforts, or both;
- 2-year peak flow/winter baseflow ratio of <20;
- greater than 60% of the upstream buffer should be greater than 30 m wide; and
- less than 15% of the sediment in the stream bed should be less than 0.85 mm.

Habitat evaluations are commonly and justifiably recognized as critical components of stream and watershed studies. However, Poole, *et al*.[108] caution users concerning their use to quantify aquatic habitat or channel morphology in an attempt to measure the response of individual streams to human activities. Their concern is the subjectivity of habitat surveys and the lack of repeatability, precision, and transferability of the measurement techniques. The measurement parameters are also assigned relatively arbitrary nominal values that are not easily statistically evaluated. They feel that the typical use of habitat unit classifications encourages the focus on direct manipulation or replacement of habitat structures (such as in stream "restoration" activities) while neglecting the long-term maintenance of habitat-forming biophysical processes (such as controlling the energy distribution of stream discharges and the discharges of sediment into the streams).

Therefore, the use of habitat unit classifications as an indicator of watershed health may be most appropriately used for only very large differences or changes, when conducted over a large portion of a watershed being studied, and only if a sufficiently large number of observations and replicates are made to compensate for the high inherent measurement variations. Many current habitat surveys are being conducted on small scales within a short period of time and with few observations, and without adequate statistical evaluations of the data. The results of these surveys are therefore of questionable value. As for all indicators, it is important that methods be developed and tested to improve the accuracy of the tool, and that additional supplemental measurement methods also be used to confirm observations and conclusions, especially when evaluating cause and effect relationships in watersheds.

Stormwater Contamination of Sediments and Increased Sediment Discharges in Urban Streams

Many of the observed biological effects associated with urban runoff may be caused by polluted sediments and associated benthic organism impacts. The EPA[109] prepared a four volume report to Congress on the incidence and severity of sediment contamination in the surface waters of the U.S. This report was required by the Water Resources Development Act of 1992. This Act defines contaminated sediment as "sediment containing chemical substances in excess of appropriate geochemical, toxicological or sediment quality criteria or measures; or otherwise considered to pose a threat to human health or the environment." In the national quality survey, the EPA examined data from 65% of the 2,111 watersheds in the U.S. and identified 96 watersheds that contain areas of probable concern. In portions of these waters, benthic organisms and fish may contain chemicals at levels unsafe for regular consumption. Areas of probable concern are located in regions affected by urban and agricultural runoff, municipal and industrial waste discharges, and other contaminant sources. When the fourth volume is completed, much more detailed information will become available concerning the relative role that urban stormwater contributes to national contaminated sediment problems. Sediment quality criteria is an emerging area, with slowly emerging general guidance available to compare locally observed conditions to "standards." In most cases, local reference conditions have been most effectively used to indicated if the observed conditions constitute a problem[13].

Examples of elevated heavy metal and nutrient accumulations in urban sediments are numerous. DePinto, *et al*.[110] found that the cadmium content of river sediments can be more than 1,000 times greater than the overlying water concentrations and the accumulation factors in sediments are closely correlated with sediment organic content. They reported that sediments were also able to adsorb phosphorus in proportion to the phosphorus concentrations in the overlaying waters during aerobic periods, but that the sediments released phosphorus during anaerobic periods. Heaney[111] found that long-term impacts of urban runoff related to the resuspension of previously deposited polluted benthos material may be more important than short-term discharges of contaminants from potential "first-flushes."

Another comprehensive study on polluted sediment was conducted by Wilber and Hunter[112] along the Saddle River in New Jersey where they found significant increases in sediment contamination with increasing urbanization. They found large variations in metal concentrations for different sediment particle sizes in the urban river. The sediment particle size distribution was the predominant influencing factor for total metal concentrations in the sediments. Areas having fine sediments had a substantially greater concentration of heavy metals than those areas having coarse sediments.

In another study, Pitt and Bozeman[22] observed concentrations for many contaminants in the urban area sediments of Coyote Creek (San Jose, California) that were much greater than those from the nonurban area. Orthophosphates, TOC, BOD_5, sulfates, sulfur, and lead were all found in higher concentrations in the sediments from the urban area stations, as compared with those from the upstream, non-urban area stations. The median sediment particle sizes were also found to be significantly smaller at the urban area stations, reflecting a higher silt content.

Several of the University of Washington projects and the Seattle METRO project investigated physical and chemical characteristics of the Kelsey and Bear Creeks sediments as part of the Bellevue, WA, NURP projects[25]. Perkins[63] found that the size and composition of the sediments near the water interface tended to be more variable and of a larger median size in Kelsey Creek than in Bear Creek. These particle sizes varied in both streams on an annual cycle in response to runoff events. Larger particle sizes were more common during the winter months when the larger flows were probably more efficient in flushing through the finer materials. Pedersen[61] also states that Kelsey

Creek demonstrated a much greater accumulation of sandy sediments in the early spring. This decreases the suitability of the stream substrates for benthic colonization. Scott, *et al*.[26] state that the level of fines in the sediment samples appears to be a more sensitive measure of substrate quality than the geometric mean of the particle size distribution. Fines were defined as all material less than about 840 microns in diameter. METRO[113] also analyzed organic priority pollutants in 17 creek sediments including several in Kelsey and Bear Creeks. Very few organic compounds were detected in either stream with the most notable trend being the much more common occurrence of various PAHs in Kelsey Creek while none were detected in Bear Creek.

Scott, *et al*.[65] state that streambed substrate quality can be an important factor in the survival of salmonid embryos. Richey[64] describes sediment bioassay tests which were performed using Kelsey and Bear Creeks sediments. She found that during the four-day bioassay experiment, no mortalities or loss of activities were observed in any of the tests. She concluded that the chemical constituents in the sediment were not acutely toxic to the test organism. However, the chronic and/or low level toxicities of these materials was not tested.

The University of Washington project and the Seattle METRO project analyzed interstitial water for various constituents. These samples were obtained by inserting perforated alu minum stand pipes into the creek sediment. This water is most affected by the sediment quality and affects in turn the benthic organisms much more than the creek water column. Scott, *et al*.[65] found that the interstitial water pH ranged from 6.5 to 7.6 and did not significantly differ between the two streams but did tend to decrease during the spring months. The lower fall temperatures and pH levels contributed to reductions in ammonium concentrations. The total ammonia and unionized ammonia concentrations were significantly greater in Kelsey Creek than in Bear Creek. They also found that the interstitial dissolved oxygen concentrations in Kelsey Creek were much below those concentrations considered normal for undisturbed watersheds. These decreased interstitial oxygen concentrations were much less than the water column concentrations and indicated the possible impact of urban development. The dissolved oxygen concentrations in the interstitial waters and Bear Creek were also lower than expected potentially suggesting deteriorating fish spawning conditions. During the winter and spring months, the interstitial oxygen concentrations appeared to be intermediate between those characteristic of disturbed and undisturbed watersheds.

The University of Washington[64] also analyzed heavy metals in the interstitial waters, focusing mostly on the more readily detected lead and zinc measurements compared to the low, or undetectable, copper and chromium concentrations. The urban Kelsey Creek interstitial water had concentrations of heavy metals approximately twice those found in the rural Bear Creek interstitial water. They expect that most of the metals were loosely bound to fine sediment particles. Most of the lead found was associated with the particulates, with very little soluble lead found in the interstitial waters. The interstitial samples taken from the stand pipe samplers were full of sediment particles which could be expected to release lead into solution following the mild acid digestion for exchangeable lead analyses. They also found that the metal concentrations in Kelsey Creek interstitial water decreased in a downstream direction. They felt that this might be caused by stream scouring of the benthic material in that part of the creek. The downstream Kelsey Creek sites were more prone to erosion and channel scouring while the most upstream station was relatively stable.

Variable interstitial water quality may cause variations in sediment toxicity with time and location. Seattle METRO[113] monitored heavy metals in the interstitial waters in Kelsey and Bear Creeks. They found large variations in heavy metal concentrations depending upon whether the sample was obtained during the wet or the dry season. During storm periods, the interstitial water and creek water heavy metal concentrations approached the stormwater values (200 μg/L for lead). During non-storm periods, the interstitial lead concentrations were typically only about 1 μg/L. They also analyzed priority pollutant organics in interstitial waters. Only benzene was found and only in the urban stream. The observed benzene concentrations in two Kelsey Creek samples were 22 and 24 μg/L, while the reported concentrations were less than 1 μg/L in all other interstitial water samples analyzed for benzene.

A number of recent investigations have examined sediment quality in conjunction with biological conditions in urban receiving waters in attempts to identify causative agents affecting the biological community. Arhelger, *et al*.[114] examined conditions in the upper Houston Ship Channel that receives drainage from the metropolitan Houston area. The channel has been dredged to allow large vessels access to the upper reaches of what used to be a relatively small channel. The dredging has increased the cross-sectional area by about 20 times, with attendant significant decreases in flushing flows. This has allowed efficient sedimentation of suspended material discharged from the 500 mi^2 urban watershed. The sediments have undergone extensive chemical, physical, and toxicity testing, with frequent indications of toxicity. The tests have indicated that the toxicity is most likely caused by the high sediment oxygen demand and associated low dissolved oxygen conditions. Toxicity testing of *Ampelisca* under varied DO conditions showed significant decreases in survival when the bottom DO is less than 3 mg/L, for example. Even though the point source BOD loads have been reduced by more than 90% since the 1970s, receiving water and sediment oxygen levels are very low, presumably caused by uncontrolled stormwater sources.

Previous studies near Auckland, New Zealand have shown that sediment concentrations of many constituents near stormwater outfalls, especially in industrial areas, often exceed guidelines intended to protect bottom-dwelling animals. Guidelines used were as presented by Long, *et al*.[115] and were as follows (along with sediment concentrations from two locations near Auckland):

mg/kg	Copper	Lead	Zinc
Effects range - low	34	47	150
Effects range - median	270	218	410
Hellyers/Kaipatiki	17 – 36	13 – 95	58 – 192
Pakuranga	14 – 65	22 – 112	108 – 345

Lead, zinc, and organochlorine were the most widespread potential problems. Field surveys and laboratory toxicity tests had shown circumstantial evidence of chronic toxicity associated with stormwater. Detailed field surveys, by Morrisey, *et al*.[116], were therefore conducted to better understand actual toxicity problems in the local marine estuaries that are influenced by complex natural factors. These complicating factors include strong gradients in salinity, sediment texture, currents, and wave action, all radically affecting the natural distribution of benthic fauna. In slowly growing areas or in relatively low density urban areas, the relatively small rate of accumulation of contaminated sediments from nonpoint sources may take many years to accumulate to levels that may produce detectable impacts in the receiving waters. In addition, changing urban conditions and changing weather from year to year make the rate of accumulation highly variable. These factors all make it difficult to conduct many types of field experiments that rely on before and after observations, or other short-term observations that assume steady conditions. They therefore relied on a "weight-of-evidence" approach considering many different and reinforcing/confirming procedures (such as the sediment quality triad and the effects range tests, both of which rely on distribution of contaminants and organisms in the field and from laboratory toxicity tests). They also applied their results to the Abundance Biomass Comparison index proposed by Warwick[117]. This index is a relative measure of biomass vs. abundance and has been shown to work well for individual sites where control sites are difficult to identify and study, especially if available "control" sites already impacted. Pore water chemistry, sediment quality, and benthic community composition were included in the field analyses. Statistical analyses identified the strongest correlations between pH and iron content of the pore water and the sediment texture, with benthic composition. The pH and iron pore water conditions may affect the bioavailability of the sediment heavy metals. Current and future work includes similar studies in non-urbanized estuaries, the development of chronic toxicity tests using local indigenous organisms, and studies of recolonization of heavily impacted sites. .

Watzin, *et al*.[118] examined sediment contamination in Lake Champlain near Burlington, VT, to compare several toxicity endpoints with sediment characteristics. They measured sediment pore water toxicity using *Ceriodaphania dubia*, *Chironomus tentans*, and *Pimephales promelas*, benthic community composition, and many physical and chemical characteristics at 19 locations. Four major storm drains and the secondary sewage treatment plant all discharged to the harbor. Boat traffic and historical petroleum handling facilities also affected some of the sampling locations. They found variable levels of toxicity at the different sites, but effects of acid-volatile sulfides on heavy metal toxicity was not demonstrated. However, they did find strong associations between metal and organic carbon levels and toxicity, indicating possible metal-organic matter complexation reducing metal availability. The sediment toxicity tests did indicate a moderate level of concern, but the macroinvertebrate community was apparently not significantly affected during these tests. They propose the use of a weight-of-evidence approach that uses multiple indicators of problems and possible sources of the problems, plus repeated observations over seasonal cycles, before management recommendations are developed.

Equilibrium partitioning of sediment-based fluoranthene and critical bioaccumulation levels was used to predict toxicity to amphipods by Driscoll and Landrum[119]. The equilibrium partitioning theory (EqP) has been used to predict effects of organic toxicants found in sediments, using an organic carbon-normalized sediment concentration of the hydrophobic organic compound (used for PAHs and pesticides) and resulting estimated pore-water concentrations. They report that toxicity bioassays with benthic invertebrates have, in general, confirmed this approach. However, certain test organisms and sediments have not been well predicted using this approach. Driscoll and Landrum tested a complementary method: the critical body residue (CBR) approach. This method measures the actual body burdens of a compound in relation to toxic effects. They found that the CBR approach is a useful complement to the EqP approach for the prediction and assessment of toxicity associated with contaminated sediments.

Rhoads and Cahill[120] studied the elevated concentrations of chromium, copper, lead, nickel and zinc that were found in sediments near storm sewer outfalls. They noted that copper and zinc concentrations were greater in the bedload compared to the bed material and therefore were more likely to be mobilized during runoff events.

Crabill, *et al*.[121] presented their analysis of the water and sediment in Oak Creek in Arizona, which showed that the sediment fecal coliform counts were on average 2200 times greater than that in the water column. Water quality standards for fecal coliforms were regularly violated during the summer due to the high recreational activity and animal activity in the watershed, as well as the storm surges due to the summer storm season.

Vollertsen, *et al*.[122] characterized the biodegradability of combined-sewer organic matter based on settling velocity. Fast settling organic matter, which represents the largest fraction of the organic material, was found to be rather slowly biodegradable compared to the slow settling organic fraction. The biodegradability of sewer sediments was argued to be taken into account for detailed characterization when dealing with CSO impacts. Vollertsen and Hvitved-Jacobsen[123] studied the stoichiometric and kinetic model parameters for predicting microbial transformations of suspended solids in combined sewer systems.

The effects of large discharges of relatively uncontaminated sediment on the receiving water aquatic environment were summarized by Schueler[124]. These large discharges are mostly associated with poorly controlled construction sites, where 30 to 300 tons of sediment per acre per year of exposure may be lost. These high rates can be 20 to 2,000 times the unit area rates associated with other land uses. Unfortunately, much of this sediment reaches urban receiving waters, where massive impacts on the aquatic environment can result. Unfortunately, high rates of sediment loss can also be associated with later phases of urbanization, where receiving water channel banks widen to accommodate the increased runoff volume and frequency of high erosive flow rates. Sediment is typically listed as one of the most important pollutants causing receiving water problems in the nations waters. Schueler[124] listed the impacts that can be associated with suspended sediment:

"• abrades and damages fish gills, increasing risk of infection and disease
• scouring of periphyton from streams (plants attached to rocks)
• loss of sensitive or threatened fish species when turbidity exceeds 25 NTU
• shifts in fish communities toward more sediment tolerant species
• decline in sunfish, bass, chub, and catfish when monthly turbidity exceed 100 NTU
• reduces sight distance for trout, with reduction in feeding efficiency
• reduces light penetration that causes reduction in plankton and aquatic plant growth
• reduces filtration efficiency of zooplankton in lakes and estuaries
• adversely impacts aquatic insects which are the base of the food chain
• slightly increases stream temperature in summer
• suspended sediments are a major carrier of nutrients and metals
• turbidity increases probability of boating, swimming, and diving accidents
• increased water treatment to meet drinking water standards

• increased wear and tear on hydroelectric and water intake equipment
• reduces anglers chances of catching fish
• diminishes direct and indirect recreational experience of receiving waters"

He also listed the impacts that can be associated with deposited sediment:

"• physical smothering of benthic aquatic insect community
• reduced survival rates for fish eggs
• destruction of fish spawning areas and redds
• 'imbedding' of stream bottom reduces fish and macroinvertebrate habitat value
• loss of trout habitat when fine sediments are deposited in spawning or riffle-runs
• sensitive or threatened darters and dace may be eliminated from fish community
• increase in sediment oxygen demand can deplete DO in lakes or streams
• significant contributing factor in the alarming decline of freshwater mussels
• reduced channel capacity, exacerbating downstream bank erosion and flooding
• reduced flood transport capacity under bridges and through culverts
• loss of storage and lower design life for reservoirs, impoundments, and ponds
• dredging costs to maintain navigable channels and reservoir capacity
• spoiling of sand beaches
• deposits diminish the scenic and recreational value of waterways"

Sediment Contamination Effects and Criteria

There is much concern and discussion about contaminated sediments in urban receiving waters. Many historical discussions downplayed the significance of contaminated sediments, based on their assumed "low-availability" to aquatic organisms. However, many of the previously described receiving water studies found greatly disturbed benthic organism populations at sites with contaminated urban sediments, compared to uncontaminated control sites. More specifically, *in-situ* sediment toxicity tests in urban receiving waters (such as those conducted by Burton and Stemmer[125]; Burton[126, 128-129]; Burton, *et al*.[127], Burton and Scott[130]; and Crunkilton, *et al*.[12]) have illustrated the direct toxic effects associated with exposure to contaminated urban sediments, to problems associated with their scour, and to decreases in toxicity associated with their removal from stormwater.

The fate of contaminated sediments, especially mechanisms that expose contaminants to sensitive organisms, can determine the overall and varied effects that the sediments may have. Scour of fine-grained sediments during periods of high flows in streams and rivers, or due to turbulence from watercraft in shallow waterbodies, has frequently been encountered. In addition, contaminant remobilization may also occur through bioturbation from sediment-dwelling organisms, or from nest-building fish. These mechanisms may resuspend contaminants, making them more available to organisms. Burrowing organisms can also transport deeply buried contaminants to surface layers, thereby increasing surface contamination levels, while the surface scouring mechanisms would tend to decrease the concentrations in the surface sediment. Bioturbation has been reported to strongly influence the fate of contaminants and that sediment-bound contaminants can be remobilized by biological activity[131].

Lee and Jones-Lee[6] reviewed the significance of chemically contaminated sediments and associated impacts. They are especially concerned about the development of sediment contamination criteria based on simple chemical tests. They feel that is has been well demonstrated that the toxic-available form of chemical constituents present in the sediment is the dissolved form present in the interstitial waters. Historically, the EPA assumed that the dissolved form of certain organic toxicants could be estimated based on an equilibrium partitioning model based on the particulate organic carbon present. Likewise, the dissolved forms of heavy metals were assumed to be controlled by metal sulfide precipitates. Lee and Jones-Lee feel that the EPA's overly simplistic two component box model used to predict dissolved forms of toxicants should never be used alone without concurrent well-established toxicity measurements. They are also concerned about the use of toxicity co-occurrence data bases used to relate measured sediment chemical conditions with observed biological conditions that are also sometimes used to establish sediment criteria. These data bases have not considered some of the most important possible causes of toxicity at the test sites, namely low dissolved oxygen, and high ammonia and hydrogen sulfide concentrations. They outlined the components of sediment toxicity tests that they feel are necessary:

- Non-chemically based "toxicity" can be caused by factors such as sediment grain size.
- Natural vs. authropogenically caused sediment toxicity also needs to be separated. They mention several instances where sediments are naturally toxic according to laboratory toxicity tests, but still support healthy and high-quality sport fisheries in overlying waters. The most obvious natural cause of sediment toxicity is low oxygen levels in the interstitial water. High levels of ammonia and hydrogen sulfide may also then occur. They state that "the presence of highly toxic conditions in sediments from natural causes, which decimates the benthic organism populations for a considerable part of the year, does not preclude the presence of an outstanding sports fishery."
- The sensitivity of the test organisms to ammonia toxicity should be considered. Several commonly used toxicity test organisms are much less sensitive to ammonia than many naturally occurring aquatic life forms of interest. Some researchers also strip ammonia from the sediments before testing, treating ammonia as a test interference. They feel that nutrient-derived toxicity (algal decomposition effects on sediment oxygen demand, and the resulting reducing conditions, low dissolved oxygen levels, and high ammonia and hydrogen sulfide levels) may be the most important cause of toxicity in aquatic sediments. An appropriate toxicity investigation evaluation (TIE) should be conducted to identify the cause of any identified toxicity problems. The use of acid volatile sulfide and heavy metal concentrations and TOC normalized sediment organic concentrations can be used as part of a TIE to rule out metals or certain organics as the potential cause of toxicity, but the reverse is not reliable (these methods cannot predict toxicity).
- Selecting reference sites is critical. A suite of test toxicity organisms (at least two or three) must be used, along with a suite of reference sites. Multiple references sites is needed to help understand the role of natural causes of toxicity. In addition, investigations should be conducted at least twice in a year during important times for the aquatic organisms.

They feel that the best approach in developing sediment quality evaluations should use a best professional judgment (BPJ), weight-of-evidence approach. This approach involves an integrated assessment of the aquatic life toxicity test results, assessment of the bioaccumulations of hazardous chemicals in edible portions of aquatic life, knowledge of chemical characteristics of the sediments and associated waters, and investigations of the aquatic life assemblages in the sediments of concern compared to appropriate reference sites.

Bioassessments and other Watershed Indicators as Components of Receiving Water Evaluations

Kuehne[132] studied the usefulness of using various aquatic organisms during stream taxonomic surveys as indicators of pollution. He found that invertebrates can reveal pollution for some time after a water pollution event, but they cannot give accurate indications of the nature of the contaminants. He stated that in-stream fish studies had not been employed as biological indicators much before 1975, but that they are comparable in many ways to invertebrates as quality indicators and can be more easily identified. However, because of better information pertaining to invertebrates and due to their limited mobility, certain species may be useful as sensitive indicators of minor changes in water quality. Fish can be highly mobile and cover large sections of a stream, as long as their passage is not totally blocked by adverse conditions. Fish disease surveys were also used during the Bellevue, Washington, urban runoff studies as an indicator of water quality problems [25, 65]. McHardy, *et al*.[133] also examined heavy metal uptake in green algae (*Cladophora glomerata*) from urban runoff for use as a biological monitor of specific metals.

Burton, *et al*.[127], during tests conducted at polluted stream and landfill sites, found that a battery of laboratory and in-situ bioassay tests were most useful when determining aquatic biota problems. The test series included microbial activity tests, along with exposures of microfaunal organisms, zooplankton, amphipods, and fathead minnows to the test water. The newly developed microbial tests correlated well with *in-situ* biological test results. Bascombe, *et al*.[134] also reported on the use of *in-situ* biological tests, using an amphipod exposed for five to six weeks in urban streams, to examine urban runoff receiving water effects. Ellis, *et al*.[135] examined bioassay procedures for evaluating urban runoff effects on receiving water biota. They concluded that an acceptable criteria for protecting receiving water organisms should not only provide information on concentration and exposure relationships for *in-situ* bioassays, but also consider body burdens, recovery rates, and sediment related effects.

A number of stormwater researchers have recently presented bioassessment and other "watershed indicators" that they have found as useful tools to quantify local receiving water problems. Many of these schemes were presented at the *Assessing the Cumulative Impacts of Watershed Development in Aquatic Ecosystems and Water Quality* conference held in Chicago in March of 1996, sponsored by the Northeastern Illinois Planning Commission, and at the *Effects of Watershed Development and Management on Aquatic Ecosystems* conference held in Snowbird, UT, in August of 1996, sponsored by the Engineering Foundation and the ASCE. Several papers from those conferences are summarized below, by location.

U.S. National Perspective of Bioassessments

Barbour[136] reviewed many of the state programs throughout the U.S. that are using biological assessments as part of their water resources programs. Most of the active state bioassessment programs started since 1990, after the publication of the EPA's *Rapid Bioassessment Protocols*[137] and the *Program Guidance for Biocriteria*[138] manuals. By 1996, numeric biocriteria were in place in Ohio and Florida (and promulgated in Maine) and under development in 13 other states. Although the majority of the states had not used biocriteria, nearly ¾ had used bioassessment data to measure the attainment of their aquatic uses. Almost all states were using benthic macroinvertebrates (all but 3 states) and fish (all but 14 states). Seven states were also using algae in their bioassessment programs.

An important aspect of the biocriteria approach is that local and regional expectations be considered in setting specific objectives. In addition, local reference sites representing specific ecoregions are also used to calibrate observations. The basic components of a bioassessment include:

- study objectives (typically the determination of biological conditions for different watershed characteristics),
- site classification (identification of homogeneous areas within a watershed, typically using various biological metrics),
- reference condition (relatively undisturbed areas for comparison and calibration of the metrics),
- standardized protocols (training and the use of consistent methods),
- data analysis (selection of several complementary metrics based on local relevancy),
- habitat assessment (physical habitat structure evaluations, generally a visual technique), and
- quality assurance (assign responsibility, establish protocols, etc. to ensure repeatability).

Watershed Indicators of Receiving Water Problems

The EPA[139] published a list of 18 indicators to track the health of the nation's aquatic ecosystems. These indicators are intended to supplement conventional water quality analyses in compliance monitoring activities. The use of broader indicators of environmental health is increasing. As an example, 12 states are currently using biological indicators, and 27 states are developing local biological indicators, according to Pelley[140]. Because of the broad nature of the nation's potential receiving water problems, this list is more general than typically used for specific stormwater issues. These 18 indicators are[139]:

1) population served by drinking water systems violating health-based requirements.
2) population served by unfiltered surface water systems at risk from microbiological contamination.
3) population served by communities by community drinking water systems exceeding lead action levels.
4) drinking water systems with source water protection programs.
5) fish consumption advisories.
6) shellfish-growing waters approved for harvest for human consumption.
7) biological integrity of rivers and estuaries.
8) species at risk of extinction.
9) rate of wetland acreage loss.
10) designated uses: drinking water supply, fish and shellfish consumption, recreation, aquatic like.
11) groundwater pollutants (nitrate).

12) surface water pollutants.
13) selected coastal surface water pollutants in shellfish.
14) estuarine eutrophication conditions.
15) contaminated sediments.
16) selected point source loadings to surface water and groundwater.
17) nonpoint source sediment loadings from crop land.
18) marine debris.

These environmental indicators cover a wide range of problems and many are for specific local uses. Most, however, are applicable to stormwater problems in urban areas.

Claytor[141, 142] summarized the approach developed by the Center for Watershed Protection as part of their EPA sponsored research on identifying watershed indicators that can be used to assess the effectiveness of stormwater management programs [143]. The indicators selected are direct or indirect measurements of conditions or elements which indicate trends or responses of watershed conditions to stormwater management activities. Categories of these environmental indicators are shown in Table 4, ranging from conventional water quality measurements to citizen surveys. Biological and habitat categories are also represented. Table 5 lists the 26 indicators, by category. It is recommended that appropriate indicators be selected from each category for a specific area under study. This will enable a better understanding of the linkage of what is done on the land, how the sources are regulated or managed, and the associated receiving water problems. The indicators were selected to: 1) measure stress or the activities that lead to impacts on receiving waters, 2) assess the resources itself, and 3) measure the regulatory compliance or program initiatives. Claytor[142] presented a framework for using stormwater indicators, as shown below:

Level 1 (Problem Identification):

1) establish management sphere (who is responsible, other regulatory agencies involved, etc.)
2) gather and review historical data
3) identify local uses which may be impacted by stormwater (flooding/drainage, biological integrity, non-contact recreation, drinking water supply, contact recreation, and aquaculture).
4) inventory resources and identify constraints (time frame, expertise, funding and labor limitations)
5) assess baseline conditions (use rapid assessment methods).

Table 4. Stormwater Indicator Categories[142]

Category	Description	Principle element being assessed
Water Quality	Specific water quality characteristics	Receiving water quality
Physical/Hydrological	Measure changes to, or impacts on, the physical environment	Receiving water quality
Biological	Use of biological communities to measure changes to, or impacts on, biological parameters	Receiving water quality
Social	Responses to surveys or questionnaires to assess social concerns	Human activity on the land surface
Programmatic	Quantify various non-aquatic parameters for measuring program activities	Regulatory compliance or program initiatives
Site	Indicators adapted for assessing specific conditions at the site level	Human activity on the land surface

Table 5. Environmental Indicators[142]

Indicator Category	Indicator Name
Water Quality Indicators	Water quality pollutant constituent monitoring Toxicity testing Non-point source loadings Exceedence frequencies of water quality standards Sediment contamination Human health criteria
Physical and Hydrological Indicators	Stream widening/downcutting Physical habitat monitoring Impacted dry weather flows Increased flooding frequency Stream temperature monitoring
Biological Indicators	Fish assemblage Macro-invertebrate assemblage Single species indicator Composite indicators Other biological indicators
Social Indicators	Public attitude surveys Industrial/commercial pollution prevention Public involvement and monitoring User perception
Programmatic Indicators	Illicit connections identified/corrected BMPs installed, inspected, and maintained Permitting and compliance Growth and development
Site Indicators	BMP performance monitoring Industrial site compliance monitoring

The selection of the indicators to assess the baseline conditions should be based on the local uses of concern, as shown on Table 6. Most of the anticipated important uses are shown to require indicators selected for each of the categories.

The Level 2 assessment strategy is for examining the local management program and is outlined below:

1) state goals for program (based on baseline conditions, resources, and constraints)
2) inventory prior and on-going efforts (including evaluating the success of on-gong efforts)
3) develop and implement management program
4) develop and implement monitoring program (more quantitative indicators than typically used for the level 1 evaluations above)
5) assess indicator results (does the stormwater indicator monitoring program measure the overall watershed health?)
6) re-evaluate management program (update and revis e management program based on measured successes and failures)

Table 6. Selection of Indicators for Evaluating Baseline Conditions, by Receiving Water Use[142]

	Water quality	Physical/ hydrological	Biological indicators	Social indicators	Programmatic indicators	Site indicators
Flooding/drainage		X		X	X	X
Biological integrity	X		X	X	X	X
Non-contact recreation	X	X	X	X	X	X
Water supply	X		X	X	X	X
Contact recreation	X	X	X	X	X	X
Aquaculture	X	X	X	X	X	X

Cave[144] described how environmental indicators are being used to summarize the massive amounts of data being generated by the Rouge River National Wet Weather Demonstration Project in Wayne County (Detroit area), MI. This massive project is examining existing receiving water problems, the performance of stormwater and CSO management practices, and receiving water responses in a 438 mi^2 watershed having more than 1.5 million people in 48 separate communities. The baseline monitoring program has now more than 4 years of continuous monitoring of flow, pH, temperature, conductivity, and DO, supplemented by automatic sampling for other water quality constituents, at 18 river stations. More than 60 projects are examining the effectiveness of stormwater management practices and 20 projects are examining the effectiveness of CSO controls, each also generating large amounts of data. Toxicants are also being monitored in sediment, water, fish tissue, and with semipermeable membranes to help evaluate human health and aquatic life effects. Habitat surveys were conducted at 83 locations along more than 200 miles of waterway. Algal diversity and benthic macroinvertebrate assessments were also conducted at these survey locations. Electrofishing surveys were conducted at 36 locations along the main river and in tributaries. Several computer models were also used to predict sources, loadings, and wet weather flow management options for the receiving waters and for the drainage systems. A geographic information system was used to manage and provide spatial analyses of the massive amounts of data collected. However, there was still a great need to simply present the data and findings, especially for public presentations. Cave described how they developed a short list of 35 indicators, based on the list of 18 from EPA and with discussions with state and national regulatory personnel. They then developed seven indices that could be color-coded and placed on maps to indicate areas of existing problems and projected conditions based on alternative management scenarios. These indices are described as follows:

Condition Quality Indicators:
1) dissolved oxygen. Concentration and % saturation values (ecologically important)
2) fish consumption index. Based on advisories from the Michigan Dept. of Public Health.
3) river flow. Significant for aquatic habitat and fish communities.
4) bacteria count. *E. coli* counts based on Michigan Water Quality Standards, distinguished for wet and dry conditions.

Multi-Factor Indices:
1) aquatic biology index. Composite index based on fish and macroinvertebrate community assessments (populations and individuals)
2) aquatic habitat index. Habitat suitability index, based on substrate, cover, channel morphology, riparian/bank condition, and water quality.
3) aesthetic index. Based on water clarity, color, odor, and visible debris.

These seven indicators represent 30 physical, chemical, and biological conditions what directly impact the local receiving water uses (water contact recreation, warmwater fishery, and general aesthetics). Cave presented specific descriptions for each of the indices and gave examples of how they are color-coded for map presentation.

The use of reference sites is common to many bioassessment approaches. As indicated above, reference sites typically are selected as representing as close to natural conditions as possible. However, it is not possible to identify such pristine locations representing varied habitat conditions in most areas of the country. Ohio, for example, has numerous reference sites throughout the state representing a broad range of conditions, but few are completely unimpacted by modifications or human activity in the watersheds. Schueler[77] reviewed a USGS report prepared by Crawford and Leant that examined the differences between streams located in forested, agricultural, and urban watersheds in North Carolina. He points out that in many cases, a completely natural forested area is not a suitable benchmark for current conditions before urbanization. In many areas of the country, agricultural land is being converted to urban land, and the in-stream changes expected may be better compared to agricultural conditions. The USGS study found that the stream impacted by agricultural operations was intermediate in quality, with higher nutrient and worse substrate conditions than the urban stream, but better macroinvertebrate and fish conditions. The forested watershed had the best conditions (good quality conditions for all categories), except for somewhat higher sediment heavy metal concentrations than expected. Even though the agricultural watershed had little impervious area, it had high sediment and nutrient discharges, plus some impacted stream corridors. The urban stream had poor macroinvertebrate and fish conditions, poor sediment and temperature conditions, and fair substrate and nutrient conditions.

Summary of Assessment Tools
Almost all states using bioassessment tools have relied on the EPA reference documents as the basis for their programs. Common components of these bioassessment programs (in general order of popularity) include:

- macroinvertebrate surveys (almost all programs, but with varying identification and sampling efforts)
- habitat surveys (almost all programs)
- some simple water quality analyses
- some watershed characterizations
- few fish surveys
- limited sediment quality analyses
- limited stream flow analyses
- hardly any toxicity testing
- hardly any comprehensive water quality analyses

Normally, numerous metrics are used, typically only based on macroinvertebrate survey results, which are then assembled into a composite index. Many researchers have identified correlations between these composite index values and habitat conditions. Water quality analyses in many of these assessments are seldom comprehensive, a possible over-reaction to conventional very costly programs that have typically resulted in minimally worthwhile information. Burton and Pitt[13] have recommended a more balanced assessment approach, using toxicity testing and carefully selected water and sediment analyses to supplement the needed biological monitoring activities. A multi-component assessment enables a more complete evaluation of causative factors and potential mitigation approaches.

Summary of Urban Runoff Effects on Receiving Waters
The effects of urban runoff on receiving water aquatic organisms or other beneficial uses is very site specific. Different land development practices create substantially different runoff flow characteristics. Different rain patterns cause different particulate washoff, transport and dilution conditions. Local attitudes also define specific beneficial uses and, therefore, current problems. There is also a wide variety of water types receiving urban runoff, and these waters all have watersheds that are urbanized to various degrees. Therefore, it is not surprising that urban runoff effects, though generally dramatic, are also quite variable and site specific.

Previous attempts to identify urban runoff problems using existing water quality data have not been conclusive because of differences in sampling procedures and the common practice of pooling data from various sites, or conditions[4]. It is therefore necessary to carefully design comprehensive, long-term studies to investigate urban runoff problems on a site-specific basis. Sediment transport, deposition, and chemistry play key roles in urban receiving waters and need additional research. Receiving water aquatic biological conditions, especially compared to unaffected receiving waters, should be studied as a supplement to laboratory bioassays. In-stream taxonomic surveys are sensitive to natural variations of pollutant concentrations, flows, and other habitat affects. However, laboratory studies are necessary to help understand potential cause and effect relationships because of their ability to better control exposure variables.

These specific studies need to examine beneficial uses directly, and not rely on published water quality criteria and water column measurements alone. Published criteria are usually not applicable to urban runoff because of the slug nature of urban runoff and the unique chemical speciation of its components. Typical natural water pollutant characteristics (especially chemical mixtures and exposure pulses) are difficult to interpret, compared to simpler artificial systems having continuous discharges of more uniform characteristics.

The long-term aquatic life effects of urban runoff are probably more important than short-term effects associated with specific events, and are related to site specific conditions associated with dilution, size of the watershed, and size of the stream. The long-term effects are probably related to habitat degradation, deposition and accumulation of toxic sediments, or the inability of the aquatic organisms to adjust to repeated exposures to high concentrations of toxic materials or high flow rates.

References

1 *Water Environment & Technology*. "News Watch: U.S. water quality shows little improvement over 1992 inventory." Vol. 8, no. 2, pp. 15 - 16. Feb. 1996a.

2 Borchardt, D. and B. Statzner. "Ecological impact of urban stormwater studied in experimental flumes: Population loss by drift and availability of refugial space." *Aquatic Sciences*. Vol. 52, No. 4. pp. 299-314. 1990.

3 Field, R., and R. Pitt. "Urban storm-induced discharge impacts: US Environmental Protection Agency research program review." *Water Science and Technology*. Vol. 22, No. 10/11. 1990.

4 Pitt, R. E. "Biological effects of urban runoff discharges." in: *Effects of Urban Runoff on Receiving Systems: An Interdisciplinary Analysis of Impact, Monitoring, and Management*, Engineering Foundation Conference, Mt. Crested Butte, CO. ASCE, NY. 1991.

5 Pitt, R. "Effects of Urban Runoff on Aquatic Biota." In: *Handbook of Ecotoxicology* (Edited by D.J. Hoffman, B.A. Rattner, G.A. Burton, Jr. and J.Cairns, Jr.). Lewis Publishers/CRC Press, Boca Raton, pp. 609-630. 1995.

6 Lee, G.F. and A. Jones-Lee. "Evaluation of the water quality significance of the chemical constituents in aquatic sediments: Coupling sediment quality evaluation results to significant water quality impacts." *WEFTEC '96, Surface Water Quality and Ecology, parts 1 and 2*. 1996 Water Environment Federation Technical Exposition and Conference, Dallas, TX. WEF, Alexandria, VA. 1996.

7 Lee, G.F. and A. Jones-Lee. "Deficiencies in stormwater quality monitoring." in: *Stormwater NPDES Related Monitoring Needs*. Edited by H.C. Torno. Proceedings of an Engineering Foundation Conference, Mt. Crested Butte, CO. August 1994. ASCE, NY. 1995.

8 Mancini, J. and A. Plummer. "Urban runoff and water quality criteria." In: *Urban Runoff Quality – Impact and Quality Enhancement Technology*. Edited by B. Urbonas and L.A. Roesner. Engineering Foundation Conference, Henniker, Hew Hampshire. ASCE, NY. pp. 133-149. June 1986.

9 Davies, P.H. "Factors in controlling nonpoint source impacts." Engineering Foundation/ASCE conference held in 1991 in Mt. Crested Butte, Colorado: *Stormwater Runoff and Receiving Systems: Impact, Monitoring and Assessment* (edited by E.E. Herricks). Lewis/CRC Press. Boca Raton. pp. 53 – 64. 1995.

10 Herricks, E.E, I. Milne, and I. Johnson. "A protocol for wet weather discharge toxicity assessment." Volume 4, pg. 13 - 24. *WEFTEC'96: Proceedings of the 69th Annual Conference & Exposition*. Dallas, Texas. 1996.

11 Herricks. E.E., editor. *Stormwater Runoff and Receiving Systems: Impact, Monitoring and Assessment.* Conference of the Engineering Foundation/ASCE held in 1991 in Mt. Crested Butte, Colorado. Lewis/CRC Press. Boca Raton. 458 pgs. 1995.

12 Crunkilton, R., J. Kleist, J. Ramcheck, W. DeVita, and D. Villeneueve. "Assessment of the response of aquatic organisms to long-term *in-situ* exposures to urban runoff." Presented at the *Effects of Watershed Developments and Management on Aquatic Ecosystems* conference. Snowbird, UT, August 4 – 9, 1996. Edited by L.A. Roesner. ASCE, New York. 1997.

13 Burton, G.A. Jr., and R. Pitt. *Stormwater Effects Handbook: A Tool Box for Watershed Managers, Scientists, and Engineers*. CRC Press, Inc., Boca Raton, FL. August 2001. 1085 pgs.

14 Duda, A.M., D.R. Lenat, and D. Penrose. "Water quality in urban streams - What we can expect." *Journal Water Pollution Control Federation*. Vol. 54, No. 7. pp. 1139- 1147. July 1982.

15 Horner, R.R. "Toward ecologically based urban runoff management." In: *Effects of Urban Runoff on Receiving Systems: An Interdisciplinary Analysis of Impact, Monitoring, and Management*. Engineering Foundation Conference. Mt. Crested Butte, CO. ASCE, NY. 1991.

16 Snodgrass, W.J., B.W. Kilgour, L. Leon, N. Eyles, J. Parish, and D.R. Barton. "Applying ecological criteria for stream biota and an impact flow model for evaluation sustainable urban water resources in southern Ontario." In: *Sustaining Urban Water Resources in the 21st Century*. Proceedings of an Engineering Foundation Conference. Edited by A.C. Rowney, P. Stahre, and L.A. Roesner. Malmo, Sweden. Sept. 7 – 12, 1997. To be published by ACSE, New York. 1998.

17 Ketchum, L.H., Jr. *Dissolved Oxygen Measurements in Indiana Streams During Urban Runoff*. EPA-600/2-78-135, U.S. Environmental Protection Agency, Cincinnati, Ohio. August 1978.

18 Heaney, J.P., W.C. Huber, M.E. Lehman. *Nationwide Assessment of Receiving Water Impacts from Urban Storm Water Pollution*. U.S. Environmental Protection Agency, Cincinnati, Ohio, April 1980.

19 Pitt, R. E. *Demonstration of Nonpoint Pollution Abatement Through Improved Street Cleaning Practices*, EPA-600/2-79-161, U.S. Environmental Protection Agency, Cincinnati, Ohio, August 1979.

20 Burkholder, J.M., L.M. Larsen, H.B. Glasgow, Jr., K.M. Mason, P. Gama, and J.E. Parsons. "Influence of sediment and phosphorus loading on phytoplankton communities in an urban piedmont reservoir." *Journal of Lake and Reservoir Management*. Vol. 14, no. 1, pp. 110 – 121. January 1998.
21 Field, R., and C. Cibik. "Urban runoff and combined sewer overflows." *Journal of Water Pollution Control Federation*. Vol. 52. No. 6. pp.1290-1307. June 1980.
22 Pitt, R. and M. Bozeman. *Sources of Urban Runoff Pollution and Its Effects on an Urban Creek*, EPA-600/52-82-090, U.S. Environmental Protection Agency, Cincinnati, Ohio, December 1982.
23 Bolstad, P.V. and W.T. Swank. "Cumulative impacts of landuse on water quality in a southern Appalachian watershed." *Journal of the American Water Resources Association*. Vol. 33, no. 3, pp. 519 – 533. June 1997.
24 Ray, S., and W. White. "Selected aquatic plants as indicator species for heavy metal pollution." *Journal of Environmental Science and Health*, A11, 717, 1976.
25 Pitt, R.E., and P. Bissonnette. *Bellevue Urban Runoff Program, Summary Report*. PB84 237213. Water Planning Division, U.S. Environmental Protection Agency, Washington, D.C., December 1983.
26 Scott, J.B., C.R. Steward, and Q.J. Stober. "Effects of urban development on fish population dynamics in Kelsey Creek, Washington." *Trans. Amer. Fisheries Society*. Vol. 115, No. 4. pp. 555-567. July 1986.
27 Marcy, S. and J. Gerritsen. "Developing deverse assessment endpoints to address multiple stressors in watershed ecological risk assessment." Abstract Book: *SETAC 17th Annual Meeting*. pg. 96. Washington, D.C., Nov. 17 - 21, 1996.
28a Dyer, S.D. and C.E. White. "A watershed approach to assess mixture toxicity via integration of public and private databases." Abstract Book: *SETAC 17th Annual Meeting*. pg. 96. Washington, D.C., Nov. 17 - 21, 1996.
28b ES&T *(Environmental Science & Technology)*. "Toxicity of aquatic mixtures yielding to new theoretical approach.". Vol. 30, no. 4, pp. 155a - 156a. April 1996a.
29 Ireland, D.S., G.A. Burton, Jr., and G.G. Hess. "In-situ toxicity evaluations of turbidity and photoinduction of polycyclic aromatic hydrocarbons." *Environmental Toxicology and Chemistry*. Vol. 15, no. 4, pp. 574 - 581. April 1996.
30 Johnson, I., E.E. Herricks, and I. Milne. "Application of a test battery for wet weather discharge toxicity analyses." Volume 4, pg. 219 - 229. *WEFTEC'96: Proceedings of the 69th Annual Conference & Exposition*. Dallas, Texas. 1996.
31 Rainbow, P.S. "Chapter 18: Heavy metals in aquatic invertebrates." *In: Environmental Contaminants in Wildlife; Interpreting Tissue Concentrations*. Edited by W.N. Beyer, G.H. Heinz, and A.W. Redmon-Norwood. CRC/Lewis Press. Boca Raton. pp. 405 - 425. 1996.
32 Allen, H.E. (editor). *Metal Contaminated Aquatic Sediments*. Ann Arbor Press. Chelsea, MI. 350 pgs. 1996.
33 Hall, L.W., Jr., M.C. Scott, W.D. Killen, Jr., and R.D. Anderson. "The effects of land-use characteristics and acid sensitivity on the ecological status of Maryland coastal plain streams." *Environmental Toxicology and Chemistry*. Vol. 15, no. 3, pp. 384 – 394. 1996.
34 Garries, M.J., T. Barron, R. Batiuk, K. Eisenman, J. Gregory, L. Hall. A. Hart, P. Jiapizian, W. Rue, J. Savitz, M.E. Setting, and C. Stoll. "Derivation of Chesapeake Bay toxics of concern." Abstract Book: *SETAC 17th Annual Meeting*. pg. 105. Washington, D.C., Nov. 17 - 21, 1996.
35 Schueler, T. (editor). "*In-situ*, nonbenthic assessment of stormwater-impacted sediments." *Watershed Protection Techniques*. Vol. 2, no. 2, pp. 351 - 353. Spring 1996b.
36 Love, E., and Woolley, B . Environmentally Sustainable Stormwater Quality. *Proc. the Eighth International Conference on Urban Storm Drainage*. August 30 – September 3, 1999, Sydney, Australia. Edited by IB Joliffe and JE Ball. The Institution of Engineers Australia, The International Association for Hydraulic Research, and The International Association on Water Quality. 50. 1999.
37 Pitt, R. "Urban stormwater toxic pollutant assessment, sources, and treatability - closure." *Water Environment Research*. Vol. 68, no. 5, pp. 953 - 955. July/Aug. 1996.
38 Huber, W.C., and Quigley, M.M . Simplified Fate and Transport Model of Runoff From Highway Construction and Repair Materials. *Proc. the Eighth International Conference on Urban Storm Drainage*. August 30 – September 3, 1999, Sydney, Australia. Edited by IB Joliffe and JE Ball. The Institution of Engineers Australia, The International Association for Hydraulic Research, and The International Association on Water Quality, 1209. 1999.
39 Doherty, F.G.; Qureshi, A.A.; and Razza, J.B . Comparison of the *Ceriodaphnia dubia* and Microtox® Inhibition Tests For Toxicity Assessment of Industrial and Municipal Wastewaters. *Environ. Toxicol*. **14**, 4, 375. 1999.

40 Kosmala, A.; Charvet, S.; Roger, M.C.; and Faessel, B . Impact Assessment of a Wastewater Treatment Plant Effluent Using Instream Invertebrates and the *Ceriodaphnia dubia* Chronic Toxicity Test. *Water Res.* (G.B.), **33**, 1, 266. 1999.
41 Marsalek, J.; Rochfort, Q.; Brownlee, B.; Mayer, T.; and Servos, M . Exploratory Study of Urban Runoff Toxicity. *Water Sci. and Tech.*(G.B.), **39**, 12, 33. 1999.
42 Marsalek J., and Rochfort, Q . Toxicity of Urban Wet-weather Pollution Sources: Stormwater and CSOs. *Proc. the Eighth International Conference on Urban Storm Drainage*. August 30 – September 3, 1999, Sydney, Australia. Edited by IB Joliffe and JE Ball. The Institution of Engineers Australia, The International Association for Hydraulic Research, and The International Association on Water Quality, 1575. 1999.
43 Skinner, L.; de Peyster, A.; and Schiff, K . Developmental Effects of Urban Storm Water in Medaka (*Oryzias latipes*) and Inland Silverside (*Menidia beryllina*). *Arch. Environ. Contam.Toxicol.*. **37**, 2, 227. 1999.
44 Tucker, K.A., and Burton, G.A . Assessment of Nonpoint-Source Runoff in a Stream Using In-Situ and Laboratory Approaches. *Environ. Toxicol. Chem.*. **18**, 12, 2797. 1999.
45 Hatch, A.C., and Burton, G.A . Sediment Toxicity and Stormwater Runoff in a Contaminated Receiving System: Consideration of Different Bioassays in the Laboratory and Field. *Chemosphere*, **39**, 6, 1001. 1999.
46 Bickford, G.; Toll, J.; Hansen, J.; Baker, E.; and Keessen, R . Aquatic Ecological and Human Health Risk Assessment of Chemicals in Wet Weather Discharges in the Sydney Region, New South Wales, Australia. *Mar. Pollut. Bull.*. **39**, 1-2, 335. 1999.
47 Brent, R.N., and Herricks, E.E . A Method for the Toxicity Assessment of Wet Weather Events. *Water Res.* (G.B.), **33**, 10, 2255. 1999.
48 Colford, J.M.; Tager, I.; Byers, L.F.; Ricci, P.; Hubbard, A.; and Horner R . Methods for Assessing the Public Health Impact of Outflows from Combined Sewer Systems. *J. Air Waste Manage. Assoc.*. **49**, 4, 454. 1999.
49 Heaney, J.P. and W.C. Huber. “Nationwide assessment of urban runoff impact on receiving water quality.” *Water Resources Bulletin*. Vol. 20. No. 1. pp. 35-42. February 1984.
50 Klein, R.D. Urbanization and Stream Quality Impairment. *Water Resources Bulletin*. Vol. 15. No. 4. August 1979.
51 Lenet, D.R., D.L. Penrose, and K. Eagleson. *Biological Evaluation of Non-Point Sources of Pollutants in North Carolina Streams and Rivers*. North Carolina Division of Environmental Management, Biological Series #102. North Carolina Dept. of Natural Resources and Community Development, Raleigh, North Carolina. 1979.
52 Lenet, D. and K. Eagleson. *Ecological Effects of Urban Runoff on North Carolina Streams*. North Carolina Division of Environmental Management, Biological Series #104. North Carolina Dept. of Natural Resources and Community Development, Raleigh, North Carolina. 1981.
53 Lenet, D.R., D.L. Penrose, and K.W. Eagleson. “Variable effects of sediment addition on stream benthos.” *Hydrobiologia*. Vol. 79. pp. 187-194. 1981.
54 Cook, W.L., F. Parrish, J.D. Satterfield, W.G. Nolan, and P.E. Gaffney. *Biological and Chemical Assessment of Nonpoint Source Pollution in Georgia: Ridge-Valley and Sea Island Streams*. Department of Biology, Georgia State University. Atlanta, Georgia. 1983.
55 Benke, A.C., G.E. Willeke, F.K. Parrish,and D.L. Stites. *Effects of Urbanization on Stream Ecosystems.* School of Biology, Environmental Resources Center, Report No. ERC 07-81. Georgia Institute of Technology. Atlanta, Georgia. 1981.
56 CTA, Inc. *Georgia Nonpoint Source Impact Assessment Study: Blue Ridge/Upland Georgia Cluster, Piedmont Cluster, and Gulf Coastal Plain Cluster*. Georgia Envir, Protection Division, Dept. of Natural Resources, Atlanta, Georgia. 1983.
57 Pratt, J.M., R.A. Coler and P.J. Godfrey. “Ecological effects of urban stormwater runoff on benthic macroinvertibrates inhabiting the Green River, Massachusetts.” *Hydrobiologia*. Vol. 83. pp. 29-42. 1981.
58 Ehrenfeld, J.G. and J.P. Schneider. *The Sensitivity of Cedar Swamps to the Effects of Non-Point Pollution Associated with Suburbanization in the New Jersey Pine Barrens*. U.S. Environmental Protection Agency. Office of Water Policy. PB8-4-136779. Washington, D.C. Sept. 1983.
59 Medeiros, C. and R.A. Coler. *A Laboratory/Field Investigation into the Biological Effects of Urban Runoff*. Water Resources Research Center, University of Massachusetts. Amherst, Massachusetts. July 1982.
60 Medeiros, C., R.A. Coler, and E.J. Calabrese. “A laboratory assessment of the toxicity of urban runoff on the fathead minnow (*Pimephales promelas*).” *Journal of Environmental Science Health*. Vol. A19. No. 7. pp. 847-861. 1984.

61 Pedersen, Edward Robert. *The Use of Benthic Invertebrate Data for Evaluating Impacts of Urban Stormwater Runoff.* Masters thesis submitted to the College of Engineering, University of Washington, Seattle, 1981.
62 Richey, Joanne Sloane, Michael A. Perkins, and Kenneth W. Malueg. "The effects of urbanization and stormwater runoff on the food quality in two salmonid streams." *Verh. Internat. Werein. Limnol.* Vol. 21, Pages 812-818, Stuttgart, October 1981.
63 Perkins, M. A. *An Evaluation of Instream Ecological Effects Associated with Urban Runoff to a Lowland Stream in Western Washington.* U.S. Environmental Protection Agency, Corvallis Environmental Research Laboratory, Corvallis, Oregon, July, 1982.
64 Richey, Joanne Sloane. *Effects of Urbanization on a Lowland Stream in Western Washington*. Doctor of Philosophy dissertation, University of Washington, Seattle, 1982.
65 Scott, J.B., C.R. Steward, and Q.J. Stober. *Impacts of Urban Runoff on Fish Populations in Kelsey Creek, Washington*. Contract No. R806387020, U.S. Environmental Protection Agency, Corvallis Environmental Research Laboratory, Corvallis, Oregon, May 1982.
66 Ebbert, J.C., J. E. Poole, and K.L. Payne. *Data Collected by the U.S. Geological Survey During a Study of Urban Runoff in Bellevue, Washington, 1979-82*. Preliminary U.S. Geological Survey Open-File Report, Tacoma, Washington, 1983.
67 Prych, Edmund A. and J.C. Ebbert. *Quantity and Quality of Storm Runoff from Three Urban Catchments in Bellevue, Washington.* Preliminary U.S. Geological Survey Water Resources Investigations Report, Tacoma, Washington, undated.
68 Mote Marine Laboratory. *Biological and Chemical Studies on the Impact of Stormwater Runoff upon the Biological Community of the Hillsborough River, Tampa, Florida*. Stormwater Management Division, Dept. of Public Works. Tampa, Florida. March 1984.
69 Striegl, R.G. *Effects of Stormwater Runoff on an Urban Lake, Lake Ellyn at Glen Ellyn, Illinois*. USGS open file report 84-603. Lakewood, Colorado. 1985.
70 Garie, H.L. and A. McIntosh. "Distribution of benthic macroinvertebrates in a stream exposed to urban runoff." *Water Resources Bulletin*, Vol. 22. No. 3. pp. 447-455. 1986.
71 Garie, H.L. and A. McIntosh. "Distribution of benthic macroinvertebrates in a stream exposed to urban runoff." *Water Science and Technology*. Vol. 22, No. 10/11. 1990.
72 Stewart, P.M.; Butcher, J.T.; and Gerovac, P.J . Diatom (*Bacillariophyta*) Community Response to Water Quality and Land Use. *Natural Areas Journal*. **19**, 2, 155. 1999.
73 Handová, Z., Z. Konícek, M. Liska, J. Marsálek, J. Matena, and J. Sed'a. "CSO impacts on receiving waters: Heavy metals in sediments and macrozoobenthos." *7th International Conference on Urban Storm Drainage*. Hannover, Germany. Sept. 9-13, 1996. Edited by F. Sieker and H-R. Verworn. IAHR/IAWQ. SuG-Verlagsgesellschaft. Hannover, Germany. pp. 485-492. 1996.
74 Boudries, H., C. Broguet, J-M. Mouchel, and D.R. Thévenot. "Urban runoff impact on composition and concentration of hydrocarbons in River Seine suspended solids." *7th International Conference on Urban Storm Drainage*. Hannover, Germany. Sept. 9-13, 1996. Edited by F. Sieker and H-R. Verworn. IAHR/IAWQ. SuG-Verlagsgesellschaft. Hannover, Germany. pp. 569-574. 1996.
75 Estèbe, A., G. Belhomme, S. Lecomte, V. Videau, J-M. Mouchel, and D.R. Thévenot. "Urban runoff impacts on particulate metal concentrations in River Seine: suspended solid and sediment transport." *7th International Conference on Urban Storm Drainage*. Hannover, Germany. Sept. 9-13, 1996. Edited by F. Sieker and H-R. Verworn. IAHR/IAWQ. SuG-Verlagsgesellschaft. Hannover, Germany. pp. 575-580. 1996.
76 Dreher, D.W. "Watershed urbanization impacts on stream quality indicators in northeastern Illinois." Presented at the *Assessing the Cumulative Impacts of Watershed Development on Aquatic Ecosystems and Water Quality* conference. March 20 – 21, 1996. Northeastern Illinois Planning Commission. pp. 129 – 135. Chicago, IL. 1997.
77 Schueler, T. (editor). "Comparison of forest, urban and agricultural streams in North Carolina." *Watershed Protection Techniques*. Vol. 2, no. 4, pp. 503 - 506. June 1997b.
78 Cordone, A.J. and D.W. Kelley. "Influences of inorganic sediments on aquatic life of streams." *Californica Fish and Game*. Vol. 47. pp. 189-228. 1961.
79 Leopold, L.B., M.G. Wolman, and J.P. Miller. *Fluvial Processes in Geomorphology*. W.H. Freeman and Co. San Francisco. 522 pgs. 1964.
80 Brookes, A. *Channelized Rivers: Perspectives for Environmental Management*. John Wiley and Sons. 1988.
81 Rosgen, D.L. *A Classification of Natural Rivers.* Catena, Elsiver Science. Amsterdam. 1994.

82 Brookes, A. "Design issues." In: *Effects of Urban Runoff on Receiving Systems: An Interdisciplinary Analysis of Impact, Monitoring, and Management*. Engineering Foundation Conference. Mt. Crested Butte, CO. ASCE, NY. 1991.
83 Wolman, M.G. and A.P. Schick. "Effects of construction on fluvial sediment, urban and suburban areas of Maryland." *Water Resources Research*. Vol. 3. No. 2. pp. 451-464. 1967.
84 Pess, G.R., and Bilby, R.E . Stream-reach and Watershed-scale Variables and Salmonid Spawning Distribution and Abundance in the Puget Sound Region. *Proc. AWRA's 1999 Annual Water Resources Conference - Watershed Management to Protect Declining Species*, December 1999, Seattle, WA. American Water Resources Association, 397. 1999.
85 Greenberg, E.S.; Gagner, M.R.; and Reiser, D.R . An Approach For Evaluating Baseline Habitat Conditions Within Urban/Urbanizing Streams With Application to Prioritization of Recovery Efforts. *Proceedings: AWRA's 1999 Annual Water Resources Conference - Watershed Management to Protect Declining Species,* December 1999, Seattle, WA. American Water Resources Association, 409. 1999.
86 Bragg, D.C., and Kershner, J.L . Coarse Woody Debris in Riparian Zones - Opportunity for Interdisciplinary Interaction. *J. Of For.*. **97**, 4, 30. 1999.
87 Larson, M.G . Effectiveness of LWD in Stream Rehabilitation Projects in Urban Basins. *Proceedings: AWRA's 1999 Annual Water Resources Conference - Watershed Management to Protect Declining Species,* December 1999, Seattle, WA. American Water Resources Association, 377. 1999.
88 Markowitz, D.V.; Gsellman, P.; and Bronowsk, J . Thinking Outside the Pipe: Non-traditional Solutions for Water Quality Improvement. *Proc. Water Environ. Fed. 72nd Annu.Conf. Exposition,* [CD-ROM], New Orleans, LA. 1999.
89 Zobrist, C.; Cencie, N.; and Demortier, G . Methodological Study of CSO's Impact: Application to the River Orne. *Proc. the Eighth International Conference on Urban Storm Drainage*. August 30 – September 3, 1999, Sydney, Australia. Edited by IB Joliffe and JE Ball. The Institution of Engineers Australia, The International Association for Hydraulic Research, and The International Association on Water Quality, 403. 1999.
90 O'Meara, J.; Murray, J.; and Ridgway, J . Restoration of an Urban Lake: the Newburgh Lake Project. *Proc. Water Environ. Fed. 72nd Annu.Conf. Exposition*, [CD-ROM], New Orleans, LA . 1999.
91 Xu, S.; Higashijima, R.; Shingu, T.; Amakata, M.; and Sakai, T . Improvement Plan of Y River Going Through Downtown of F City. *Proc. the Eighth International Conference on Urban Storm Drainage*. August 30 – September 3, 1999, Sydney, Australia. Edited by IB Joliffe and JE Ball. The Institution of Engineers Australia, The International Association for Hydraulic Research, and The International Association on Water Quality, 218. 1999.
92 Cianfrani, C.; Hession, W.C.; McBride, M.; and Pizzuto, J.E . Urban Streams in Fairmount Park, Philadelphia, PA: Assessment and Restoration. *Proceedings: AWRA's 1999 Annual Water Resources Conference - Watershed Management to Protect Declining Species*, December 1999, Seattle, WA. *Am. Water Resour. Assoc.*, 393. 1999.
93 Derry, W.E.; Haub, A.; Dobey, E.; and Hoey, J . City of Olympia Aquatic Habitat Evaluation and Management Study. *Proceedings: AWRA's 1999 Annual Water Resources Conference - Watershed Management to Protect Declining Species*, December 1999, Seattle, WA. American Water Resources Association, 323. 1999.
94 Ishikawa, T.; Irie, M.; Terashima, T.; and Ozawa, K . Restoration of the Hydrological Cycle in the Izumi River Basin in Yokohama. *Proc. the Eighth International Conference on Urban Storm Drainage*. August 30 – September 3, 1999, Sydney, Australia. Edited by IB Joliffe and JE Ball. The Institution of Engineers Australia, The International Association for Hydraulic Research, and The International Association on Water Quality, 530. 1999.
95 Saeki, K.; Houjoh, Y.; Inoyae, I.; Shinoda, Y.; Watanabe, S.; and Nakanishi, Y . Rehabilitation of the Water Cycle at the Basin of the Kanda River. *Proc. the Eighth International Conference on Urban Storm Drainage*. August 30 – September 3, 1999, Sydney, Australia. Edited by IB Joliffe and JE Ball. The Institution of Engineers Australia, The International Association for Hydraulic Research, and The International Association on Water Quality, 522. 1999.
96 Kennen, J.G . Relation of Macroinvertebrate Community Impairment to Catchment Characteristics in New Jersey Streams. *J. Am. Water Resour. Assoc.*. **35**, 4, 939. 1999.
97 Bhaduri, B., M.Grove, C. Lowry and J. Harbor. "Assessing long-term hydrologic effects of land use change." *Journal AWWA*. Vol. 89, No. 11, pg. 94 – 106. Nov. 1997.

98 Booth, D.B., C.R. Jackson. "Urbanization of aquatic systems: Degradation thresholds, stormwater detection, and the limits of mitigation." *Journal of the American Water Resources Association*. Vol. 33, no. 5, pp. 1077 – 1090. October 1997.

99 Schueler, T. (editor). "Stream channel geometry used to assess land use impacts in the Pacific Northwest." *Watershed Protection Techniques*. Vol. 2, no. 2, pp. 345 - 348. Spring 1996a.

100 Stephenson, D. "Evaluation of effects of urbanization on storm runoff." *7th International Conference on Urban Storm Drainage*. Hannover, Germany. Sept. 9-13, 1996. Edited by F. Sieker and H-R. Verworn. IAHR/IAWQ. SuG-Verlagsgesellschaft. Hannover, Germany. pp. 31-36. 1996.

101 Maxted, J.R. "The use of percent impervious cover to predict the ecological condition of wadable nontidal streams in Delaware." Presented at the *Assessing the Cumulative Impacts of Watershed Development on Aquatic Ecosystems and Water Quality* conference. March 20 – 21, 1996. Northeastern Illinois Planning Commission. pp. 123 – 127. Chicago, IL. 1997.

102 MacRae, C.R. "Experience from morphological research on Canadian streams: Is control fo the two-year frequency runoff event the best basis for stream channel protection?" Presented at the *Effects of Watershed Developments and Management on Aquatic Ecosystems* conference. Snowbird, UT, August 4 – 9, 1996. Edited by L.A. Roesner. ASCE, New York, 1997.

103 Sovern, D.T. and P.M. Washington. "Effects of urban growth on stream habitat." Presented at the *Effects of Watershed Developments and Management on Aquatic Ecosystems* conference. Snowbird, UT, August 4 – 9, 1996. Edited by L.A. Roesner. pp. 163 – 177. ASCE, New York, 1997.

104 EPA (U.S. Environmental Protection Agency). *Results of the Nationwide Urban Runoff Program*. Water Planning Division, PB 84-185552, Washington, D.C., December 1983.

105 Booth, D.B., D.R. Montgomery, and J. Bothel. "Large woody debris in urban streams of the Pacific Northwest" Presented at the *Effects of Watershed Developments and Management on Aquatic Ecosystems* conference. Snowbird, UT, August 4 – 9, 1996. Edited by L.A. Roesner. ASCE, New York, 1997.

106 Montgomery, D.R. and J.M. Buffington. *Channel Classification, Prediction of Channel Response, and Assessment of Channel Conditions*. Washington State Department of Natural Resources. Report TFW-SH10-93-002. 84 pgs. 1993.

107 Horner, R.R, D.B. Booth, A. Azous, and C.W. May. "Watershed determinants of ecosystem functioning." Presented at the *Effects of Watershed Developments and Management on Aquatic Ecosystems* conference. Snowbird, UT, August 4 – 9, 1996. Edited by L.A. Roesner. ASCE, New York, 1997.

108 Poole, G.C., C.A. Frissell, and S.C. Ralph. "In-stream habitat unit classification: Inadequacies for monitoring and some consequences for management." *Journal of the American Water Resources Association*. Vol. 33, no. 4, pp. 879 – 896. August 1997.

109 EPA (U.S. Environmental Protection Agency). Report to Congress: *The Incidence and Severity of Sediment Contamination in Surface Waters of the United States*, Volume 1: *National Sediment Quality Survey* (EPA 823-R-97-006), Volume 2: *Data Summary for Areas of Probable Concern (APC)* (EPA 823-R-98-007), Volume 3: *National Sediment Contaminant Point Source Inventory* (EPA 823-R-98-008), Volume 4 (under development): *National Sediment Contaminant Nonpoint Source Inventory*. National Center for Environmental Publications and Information. Cincinnati, Ohio. 1998.

110 DePinto, J.V., T.C. Young and S.C. Martin. "Aquatic sediments." *Journal of Water Pollution Control Federation*. Vol. 52. No. 6. pp 1656-70. June 1980.

111 Heaney, J.P. *Nationwide Assessment of Receiving Water Impacts from Urban Storm Water Pollution: First Quarterly Progress Report*. Environmental Engineering Sciences, University of Florida, Gainesville, Florida, 1978.

112 Wilber, W. G., and J. V. Hunter. *The Influence of Urbanization on the Transport of Heavy Metals in New Jersey Streams*, Water Resources Research Institute, Rutgers University, New Brunswick, New Jersey, 1980.

113 Galvin, David D. Richard K. Moore. *Toxicants in Urban Runoff, METRO Toxicant Program Report #2*. U.S. Environmental Protection Agency Grant #P-000161-01, Lacey, Washington, December, 1982.

114 Arhelger, M., P. Jensen. Amd Y-C. Su. "Houston ship channel sediments and their relation to water quality." Volume 4, pg. 329 - 337. *WEFTEC'96: Proceedings of the 69th Annual Conference & Exposition*. Dallas, Texas. 1996.

115 Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. "Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments." *Environmental Management*. Vol. 19, pp. 81 – 97. 1996.

116 Morresey, D.J., D.S. Roper, and R.B. Williamson. "Biological effects of the build-up of contaminants in sediments in urban estuaries." Presented at the *Effects of Watershed Developments and Management on Aquatic Ecosystems* conference. Snowbird, UT, August 4 – 9, 1996. Edited by L.A. Roesner. ASCE, New York, pp. 1 – 20. 1997.
117 Warick, R.M. "A new method for detecting pollution effects on marine macrobenthic communities." *Marine Biology*. Vol. 92, pp. 557 – 562. 1986.
118 Watzin, M.C., A.W. McIntosh, E.A. Brown, R. Lacey, D.C. Lester, K.L. Newbrough, and A.R. Williams. "Assessing sediment quality in heterogeneous environments: A case study of a small urban harbor in Lake Champlain, Vermont, USA." *Environmental Toxicology and Chemistry*. Vol. 16, No. 10. pp. 2125 – 2135. 1997.
119 Driscoll, S.K., and P.F. Landrum. "A comparison of equilibrium partitioning and critical body residue approaches for predicting toxicity of sediment-associated fluoranthene to freshwater amphipods." *Environmental Toxicology and Chemistry*. Vol. 16, no. 10, pp. 2179 – 2186. 1997.
120 Rhoads, B.L., and Cahill, R.A . Geomorphological Assessment of Sediment Contamination in an Urban Stream System. *Applied Geochemistry*. **14**, 4, 459. 1999.
121 Crabill, C.; Donald, R.; Snelling, J.; Foust, R.; and Southam, G . The Impact of Sediment Fecal Coliform Reservoirs on Seasonal Water Quality in Oak Creek, Arizona. *Water Res.* (G.B.), **33**, 9, 2163. 1999.
122 Vollertsen, J.; Hvitved-Jacobsen, T.; McGregor, I.; and Ashley, R. (1999b) Aerobic Microbial Transformations of Pipe and Silt Trap Sediments from Combined Sewers. *Water Sci. Technol.* (G.B.). **39**, 2, 233. 1999.
123 Vollertsen , J., and Hvitved-Jacobsen, T . Stoichiometric and Kinetic Model Parameters for Microbial Transformations of Suspended Solids in Combined Sewer Systems. *Water Res.* (G.B.). **33,**14, 3127. 1999.
124 Schueler, T. (editor). "Impact of suspended and deposited sediment." *Watershed Protection Techniques*. Vol. 2, no. 3, pp. 443. February 1997a.
125 Burton, G.A., Jr., B.L. Stemmer, and K. L. Winks. "A multitrophic level evaluation of sediment toxicity in Waukegan and Indiana Harbors," *Environ. Toxicol. Chem.* 8:1057-1066. 1989.
126 Burton, G.A., Jr. "Evaluation of seven sediment toxicity tests and their relationships to stream parameters." *Toxicity Assess.* 4:149-159. 1989.
127 Burton, G.A., Jr., B.L. Stemmer, and K. L. Winks. "A multitrophic level evaluation of sediment toxicity in Waukegan and Indiana Harbors," *Environ. Toxicol. Chem.* 8:1057-1066. 1989.
128 Burton, G.A., Jr. "Assessing freshwater sediment toxicity." *Environ. Toxicol. Chem.* 10(12).1991.
129 Burton, G.A., Jr., Ed. *Sediment Toxicity Assessment.* Boca Raton, FL. Lewis Publishers. 1992.
130 Burton, G.A., Jr., and J. Scott. "Sediment toxicity evaluations," *Environ. Sci. Technol.* Vol. 25. 1992.
131 ES&T *(Environmental Science & Technology)*. "News –Sediments: Pollutant remobilization." Vol. 31, no. 12, pg. 536a. 1997.
132 Kuehne, R.A. *Evaluation of Recovery in a Polluted Creek After Installment of New Sewage Treatment Procedures*. University of Kentucky Water Resources Research Institute, Lexington, Kentucky. May 1975.
133 McHardy, B.M., J.J. George, and J. Salanki (editors). "The uptake of selected heavy metals by the green algae *Cladophora glomerata*." In: *Proceedings of Symposium on Heavy Metals in Water Organisms*. Tihany, Hungary. Akademiai Kiado, Budapest, Hungary. *Symp. Biol. Hung*., Vol. 29. pp. 3-20. 1985.
134 Bascombe, A.D., J.B. Ellis, D.M. Revitt, and R.B.E. Shutes. "Development of ecotoxicological criteria in urban catchments." *Water Science and Technology*. Vol. 22, No. 10/1, pp. 173-179. 1990.
135 Ellis, J.B., R.B. Shutes, and D.M. Revitt. "Ecotoxicological approaches and criteria for the assessment of urban runoff impacts on receiving waters." In: *Effects of Urban Runoff on Receiving Systems: An Interdisciplinary Analysis of Impact, Monitoring, and Management*. Engineering Foundation Conference. Mt. Crested Butte, CO. ASCE, NY. 1991.
136 Barbour, M.T. "Measuring the health of aquatic ecosystems using biological assessment techniques: A national perspective." Presented at the *Effects of Watershed Developments and Management on Aquatic Ecosystems* conference. Snowbird, UT, August 4 – 9, 1996. Edited by L.A. Roesner. ASCE, New York, pp. 18 – 33. 1997.
137 Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. *Rapid Bioassessment Protocols for use in Streams and Rivers: Benthic Macroinvertebrates and Fish*. EPA-440-4-89-001. U.S. Environmental Protection Agency. Office of Water Regulations and Standards. Washington, D.C. 1989.
138 EPA (U. S. Environmental Protection Agency). *Biological Criteria: National Program Guidance for Surface Waters*. EPA-440-5-90-004. U.S. Environmental Protection Agency. Office of Water Regulations and Standards. Washington, D.C. 1990.

139 EPA (U.S. Environmental Protection Agency). *Environmental Indicators of Water Quality in the United States*. Office of Water, U.S. Environmental Protection Agency. EPA 841-F-96-002. Washington, D.C., June 1996.

140 Pelley, J. “National ‘environmental indicators’ issued by EPA to track health of U.S. waters.” *Environmental Science & Technology*, Vol. 31, no. 9, pg. 381a. Sept. 1996.

141 Claytor, R.A. “An introduction to stormwater indicators: an urban runoff assessment tool.” *Watershed Protection Techniques*. Vol. 2, no. 2, pp. 321 - 328. Spring 1996a.

142 Claytor, R.A. “An introduction to stormwater indicators: Urban runoff assessment tools.” Presented at the *Assessing the Cumulative Impacts of Watershed Development on Aquatic Ecosystems and Water Quality* conference. March 20 – 21, 1996. Northeastern Illinois Planning Commission. pp. 217 – 224. Chicago, IL. April 1997.

143 Claytor, R.A. and W. Brown. *Environmental Indicators to Assess the Effectiveness of Municipal and Industrial Stormwater Control Programs*. Prepared for the U.S. EPA, Office of Wastewater Management. Center for Watershed Protection, Silver Spring, MD. 210 pgs. 1996.

144 Cave, K.A., “Receiving water quality indicators for judging stream improvement.” In: *Sustaining Urban Water Resources in the 21st Century*. Proceedings of an Engineering Foundation Conference. Edited by A.C. Rowney, P. Stahre, and L.A. Roesner. Malmo, Sweden. Sept. 7 – 12, 1997. To be published by ACSE, New York. 1998.

145 Pitt, R., M. Lilburn, S. Nix, S.R. Durrans, S. Burian, J. Voorhees, and J. Martinson *Guidance Manual for Integrated Wet Weather Flow (WWF) Collection and Treatment Systems for Newly Urbanized Areas (New WWF Systems).* U.S. Environmental Protection Agency. 612 pgs. Expected publication in 2001.

146 Heaney, J.P., R. Pitt, and R. Field. *Innovative Urban Wet-Weather Flow Management Systems*. ISBN 1-56676-914-0. Technomics, Lancaster, PA. 535 pgs. June 2000.

147 Burton, G.A., Jr., R. Pitt, and S. Clark. “The role of whole effluent toxicity test methods in assessing stormwater and sediment contamination.” *CRC Critical Reviews in Environmental Science & Technology*. 30: 413-447. 2000.