

Watershed Processes and Aquatic Resources: A Literature Review

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Introduction

The landscape is a dynamic place. Rain falls; snow melts; water moves across the land and seeps into the ground. Vegetation grows, producing foliage. Leaf litter and downed wood decompose, providing food and building soils. The landscape is constantly changing, sometimes catastrophically with landslides and floods, but most often at speeds too slow, or at scales too small, to observe directly.

The ways that energy and materials are produced, stored, and move across the land are processes that occur naturally, and have been the primary drivers that have shaped the development of habitat. In turn, human development of the land can significantly alter the magnitude and timing of the processes themselves. It is also true that these alterations can cause changes in habitat formation and stability downstream and downslope from the original disturbance. Most commonly, this effect is mediated by hydraulically driven processes: the delivery and routing of water, sediment, large and small wood, nutrients and toxicants.

Disturbance regimes

One way to look at ecosystem processes is through the lens of disturbance regimes. As noted above, disturbance arising from the movement of energy and material through the landscape plays a major role in maintaining diversity in many ecosystems (Cederholm et al. 2001). Observable changes can range from common, low intensity events, such as seasonal changes in stream flow, to less frequent, higher intensity disturbances, such as large fires (Cederholm et al. 2001). Natural disturbances are often modified or prevented in areas where humans have settled (Chappell et al. 2001). A major difference between urbanized and undeveloped communities is often a decrease in the frequency of disturbance events (Chappell et al. 2001).

Except in coastal forests, fire is one of the most important and widespread disturbance factors, especially east of the Cascade Crest (Johnson and O'Neil 2001, McComb 2001).

Because of the hazards fire poses to human life and structures, fire is most often suppressed in areas where people live. Within urban areas fire is usually caused by human accident or arson. The loss of a natural fire regime has varied effects on native ecosystems. Plant species composition can be altered where the understory is built up through fire suppression. This increased biomass, in turn, can become additional fuel that can make fires burn more intensely (Sallabanks et al. 2001). Suppression can also diminish habitat for reptiles and is known to alter the distribution and abundance of some avian species (Sallabanks et al. 2001).

Another common disturbance event is flooding. In urban areas, the flood regime is often altered by channelization of streams, diking, and paving. Floods in developing landscapes are now more frequent and more intense than those experienced before the modifications were constructed (Chappell et al. 2001).

Other common types of natural disturbances in Washington include tree blowdown, landslides, droughts, and disturbances associated with certain animals (e.g., beaver, ground squirrels; Chappell et al. 2001). Changes in abundance of animals that burrow, build dams, and trample vegetation also represent changes in the disturbance regimes they support.

Delivery and routing of water

Urbanization alters the hydrologic regime of surface waters by changing the way water cycles through a drainage basin. In a natural setting, precipitation is intercepted or delayed by the forest canopy and ground cover. Vegetation, depressions on the land, and soils provide extensive storage capacity for precipitation. Water exceeding this capacity travels via shallow subsurface flow and groundwater and eventually discharges gradually to surface water bodies. In a forested, undisturbed watershed, direct surface runoff occurs rarely or not at all because precipitation intensities do not exceed soil infiltration rates. Figures 1 and 2 illustrate this shift in hydrologic regime.

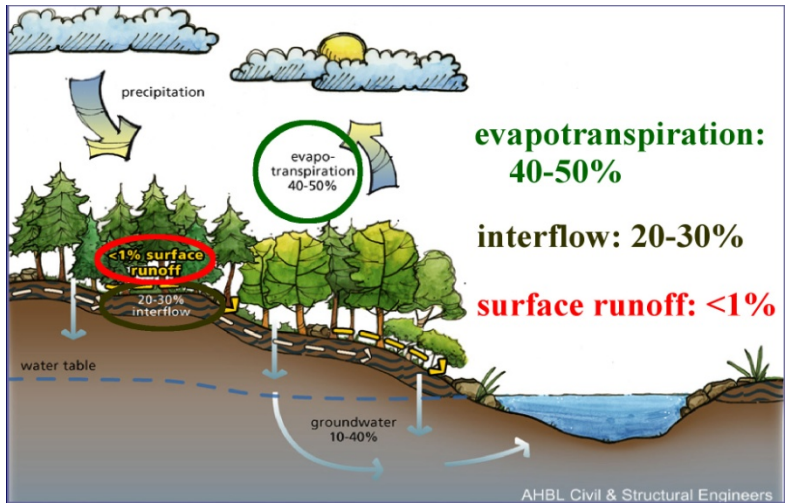


Figure 1 – Natural Hydrologic Regime(from 2005 PSAT LID Manual).

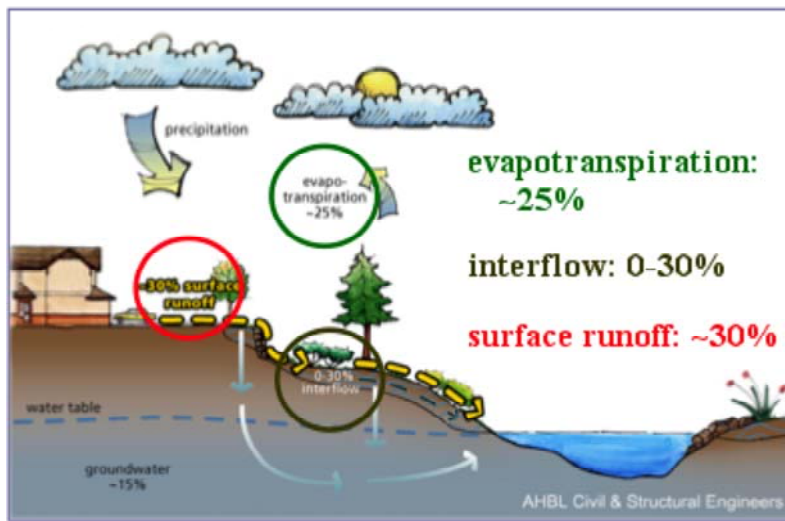


Figure 2 – Urbanized Hydrologic Regime (from 2005 PSAT LID Manual)

During the initial phases of urbanization, clearing of native vegetation reduces or eliminates interception storage and the water reservoir in soils. Loss of vegetation and “duff” (mostly composting vegetative material) from the understory takes away another storage reservoir. Site grading eliminates natural depressions. Impervious surfaces, of course, stop any infiltration and produce surface runoff. Even when surfaces remain pervious, construction often removes, erodes, or compacts topsoil. The compacted, exposed soil retards infiltration and offers much less storage capacity. Development typically replaces natural drainage systems with hydraulically efficient pipe or ditch networks that shorten the travel time of runoff to the receiving water (Hirsch et al. 1990).

The many changes brought on by urbanization tend to alter streamflow patterns in characteristic ways. Figure 3 illustrates typical hydrographs (flow rate versus time) for a stream before and after watershed urbanization. The hydrograph emphasizes the higher peak flow rate of urbanized basins compared to natural landscape conditions. The area under the hydrograph curves represents the total runoff volume, which is significantly greater for the urbanized condition. In addition, when more impervious surfaces exist there is typically less delay between rainfall and runoff. The construction of an engineered stormwater drainage network also invariably increases the drainage density of urbanizing basins (Graf 1977). Typically, these engineered conveyance systems are designed specifically to remove water from the natural drainage network and so reduce the time necessary for overland flow to reach stream channels. The net effect of these urban watershed changes is that a higher proportion of rainfall is translated into runoff, which occurs more rapidly, and the resultant flood flows are therefore higher and much more “flashy” than natural catchments (Hollis 1975).

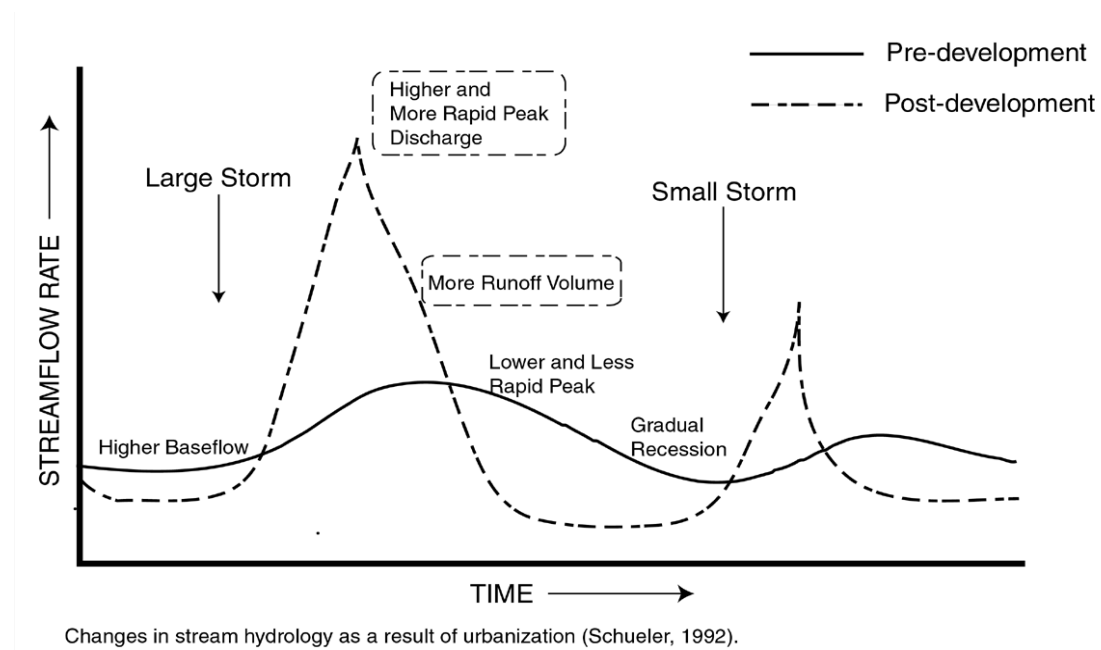


Figure 3 – Streamflow Hydrographs.

In general, the hydrologic changes associated with urbanization can be traced primarily to the loss of natural land cover (vegetation and soil) and the increase in impervious surfaces in the watershed (Dunne and Leopold 1978). The impact of urbanization and impervious surfaces on watershed hydrology has been studied for many decades. Wilson (1967) looked at the impact of urbanization on flooding in Jackson, Mississippi. Early research by Leopold (1968) reported that a two- to five-fold increase in peak streamflow was common in urbanizing basins, although some streams showed an even greater rise, especially in arid areas. Seaburn (1969) studied the effects of urbanization on stormwater runoff on Long Island, New York, finding similar results. Hammer (1973)

also found that peak streamflows increased with greater watershed urbanization.

A decline in groundwater recharge is also common in urbanizing watersheds, due to greater impervious areas and less infiltration (Foster et al. 1994). Bhaduri et al. (1997) also quantified the changes in streamflow and related decreases in groundwater recharge associated with watershed urbanization in the Midwest.

A study from western Washington illustrates these changes in hydrologic function that occur during the development process (Burgess et al. 1998). To estimate the hydrologic balance for two basins in close proximity, an approach was used combining hydrologic modeling and simple monitoring. At the time of the study, both basins were in suburban areas, but one was relatively undeveloped, while the other was suburban in land use. Before being developed, the Novelty Hill and Klahanie basins were hydrologically similar. Both study basins are in the same geological region and were once largely forested. Novelty Hill was significantly deforested, and 30 percent of the area was covered with impervious surfaces. In this study, Novelty Hill had a faster flow response, higher peak flow, and longer time of discharge. Also, there was more flow response when there was preceding wetness in the soil. For the annual water balance in this basin (the difference between precipitation and catchment outflow), 69 to 88 percent of annual precipitation left as groundwater recharge or evapo-transpiration (Burgess et al. 1998). Because the soil at Klahanie is deeper and less disturbed than at Novelty Hill, it takes more precipitation to saturate. In the developed Novelty Hill basin, 44 to 48 percent of the annual precipitation left as catchment outflow, as opposed to about 12 to 30 percent in Klahanie (Burgess et al. 1998). One of the most interesting findings of this study was that runoff from what are considered pervious areas such as lawns and landscaped areas accounted for 40 to 60 percent of the total annual runoff in the developed basin (Burgess et al. 1998). In addition, the loss of local depressional storage likely influences hydrologic function of lawns and landscaped areas converted from natural forested areas. This study also illustrates that imperviousness encompasses much more than just paved surfaces.

As mentioned in the previous section, watershed urbanization also has an impact on flooding frequency, duration, and intensity. Hollis (1975) studied the impact of urbanization on flood recurrence interval. This research found that, in general, floods with a return period of one year or longer are not affected by a watershed impervious level of approximately 5 percent. In addition, small flooding events and peak streamflows may be increased by up to 10 times that found under natural conditions. Hollis (1975) also found that under typical (~30 percent imperviousness) urbanized conditions, 100-year floods are doubled in magnitude due to the greater runoff volume. Finally, the hydrologic effect of urbanization tends to decline, in relative terms, as flood recurrence intervals increase (Hollis 1975). As a result, in urbanized watersheds, it is not uncommon for a flood event with a 10-year recurrence interval to shift to a more frequent 2-year interval (Hollis 1975). In addition, the discharge rates of small, frequent

floods tend to increase by a greater percentage of pre-development rates than those of large, infrequent floods (Hollis 1975).

As a result of watershed development, the frequency of bankfull flows can also be significantly increased in urbanizing stream basins. In western Washington, a computer model capable of continuous simulation was used to study the hydrology of two similar watersheds (Booth 1991). This study compared a fully forested basin with a developed (approximately 40 percent impervious area) basin. The model predicted that the pre-development discharge that occurs only once in five years would occur in 39 of 40 years after urbanization, essentially on an annual basis. These alterations in hydrologic characteristics can result in a significant change in the disturbance regime of a typical stream ecosystem (Booth 1991).

In a study in the Toronto area of Ontario, Canada (Snodgrass et al. 1998), the bankfull streamflow recurrence period was 1.5 years under natural conditions. Storms that result in bankfull flows were generally found to be in equilibrium with the natural resisting forces (e.g., stream bank vegetation) that tend to stabilize the stream channel. As watersheds urbanized, bankfull flows occurred more frequently, up to about every 0.4 years within Toronto (Snodgrass et al., 1998). A study in the upper Accotink Creek watershed in northern Virginia related the increase in impervious surface area from development to changes in streamflow over the period 1949 to 1994 (Jennings and Jarnagin 2002). Over this period, total impervious area (TIA) increased from 3 percent to 33 percent. Over the same period, the response of streamflow discharge to precipitation events increased significantly, as did the frequency of peak events (Jennings and Jarnagin 2002). Other studies have shown similar results. In a stream study in Washington State, the flow rate that had been reached only once in 10 years on average before development, increased in frequency to about every two years after urbanization (Scott 1982). In a similar study in Korea, the peak discharge of runoff increased and the mean lag time of the study stream decreased over a period of two decades, due to increasing urbanization (Kang et al. 1998).

Another important characteristic of highly impervious, urbanized watersheds is the production of runoff during even relatively small storm events. Under natural conditions, small precipitation events generally produce little, if any, runoff. This is due to the interception and evapo-transpiration of rainfall by native vegetation as well as to the absorption of rainfall by the upper soil horizon and rainfall held in natural depressions, where it eventually infiltrates or evaporates. It has been estimated that the residence time of precipitation in natural depressional storage is typically at least 4 times that of impervious surfaces (Novotny and Chesters 1981). A study in Australia found that the average peak discharge for urban streams was 3.5 times higher than the peak flow for rural streams (Neller 1988).

Booth (1991) noted that in addition to high-flow peaks being amplified in urban stream

hydrographs in the Puget Sound region, new peaks also appeared. These new peaks were the result of small storms, most of which produced no runoff under pre-development conditions but generated substantial flows under the urbanized condition. Therefore, it can be concluded that watershed development does more than just magnify peak flows and flooding events; it also creates entirely new high-flow events due to runoff from impervious surfaces.

Yet another characteristic of urban streams is the more rapid recession of stormflow peaks. In addition, the baseflow conditions in urban streams are typically lower in urbanized watersheds. This has been observed for wet season baseflows in the Puget Sound region (Konrad and Booth 2002) and in the Chesapeake Bay region (Klein 1979). In arid areas, there may also be a noticeable decrease in dry season baseflow due to watershed development (Harris and Rantz 1964). A study in Long Island, New York revealed the extent of seasonal hydrologic shifts in urban streams. In several undeveloped watersheds, stream baseflow constituted up to 95 percent of annual discharge. That proportion dropped to 20 percent after development (Simmons and Reynolds 1982).

Rose and Peters (2001) examined streamflow characteristics that changed during the period from 1958 to 1996 in a highly urbanized watershed (Peachtree Creek), compared to less urbanized watersheds and non-urbanized watersheds, in the vicinity of Atlanta, Georgia. Data was obtained from seven U.S. Geological Survey (USGS) stream gages, 17 National Weather Service rain gages, and five USGS monitoring wells. The fraction of the rainfall occurring as runoff in the urban watershed was not significantly greater than that in the less urbanized watersheds, but this ratio did decrease from the higher elevation and higher relief watersheds to those with lower elevation and lower relief. For the 25 largest storms, the peak flows for the urban creek were 30 to 100 percent greater than the peak flows in the streams located in the less developed areas. In the urban stream, the flow rates also decreased more rapidly after storms than in the other streams. The low flow in the urban creek was 25 to 35 percent lower than in the less developed streams, likely caused by decreased infiltration due to the more efficient routing of stormwater and the paving of groundwater recharge areas.

In an extensive stream research project in Wisconsin, the observed decrease in stream baseflow was strongly correlated with watershed imperviousness (Wang et al. 2001). Similarly, an urban stream study in Vancouver, British Columbia, Canada, monitored eleven urbanizing small-stream watersheds. Baseflow and groundwater recharge were consistently lower in watersheds with more than 40 percent impervious cover (Finkebine et al. 2000). Both of these studies found linkages between these shifts in hydrologic regime and both habitat degradation and the decline in biological integrity in the urbanizing streams.

Sheeder and others (2002) investigated the hydrograph responses to dual rural and

urban land uses in three small watersheds. Two important conclusions were deduced from this investigation. First, in all cases, the researchers found two distinct peaks in stream discharge, each representing different contributing areas to direct discharge with greatly differing curve numbers and lags, representative of urban and rural source regions. Second, the direct discharge represented only a small fraction of the total drainage area, with the urban peak becoming increasingly important in relation to the rural peak as urbanization increases and the magnitude of the rain event decreases.

Nagasaka and Nakamura (1999) examined the influences of land-use changes on the hydrologic response and the riparian environment in a northern Japanese area. Temporal variance in a hydrological system and riparian ecosystem was examined with reference to land-use conversion in order to clarify the linkages between the two. The results indicated that the hydrological system had been altered since the 1970s, with flood peaks increased by a factor of 1.5 to 2.5. The time when peak flow appeared was shortened by seven hours. The ecological systems were closely related to and distinctly altered by the changes that had occurred in local land use. A similar study in southern California found comparable results (White and Greer 2002).

Adjacent to water bodies, floodplain encroachment eliminates another storage zone needed to diminish high flows. When the channel cannot contain the greater flow, flooding results. Clearing riparian vegetation removes wood that helps slow down the flow and, in many cases, prevent bed and bank erosion. Clearing also eliminates shade, refuge, and food supply. Urban residents and high streamflows remove remaining wood, further decreasing the stream's opportunity to dissipate energy without flooding or damaging the channel (Dunne and Leopold 1978). In addition, any channel modifications (e.g., streambank armoring, levee construction, or diking) that inhibit stream-floodplain interactions can have serious consequences for downstream flooding.

Biological and ecological effects of urban hydrologic change

As discussed above, the hydrologic impacts of watershed urbanization include the following:

- Greater runoff volume from impervious surfaces
- Higher flood recurrence frequency
- Less lag time between rainfall, runoff, and streamflow response
- Higher peak streamflow for a given size storm event
- More bankfull or higher streamflows – flashier flows
- Longer duration of high streamflows during storm events
- More rapid recession from peak flows
- Lower wet and dry season baseflow levels
- Less groundwater recharge
- Greater wetland water level fluctuation

All of these characteristics represent alterations in the natural hydrologic regime to which aquatic biota have adapted over the long term. These are significant hydrologic changes that can negatively impact aquatic biota directly or indirectly. Direct impacts include washout of organisms from their preferred habitat and the physiological stress of swimming in higher flows. Indirect impacts are centered on the degradation of instream habitat that occurs as a result of the higher urban streamflows. These higher flows result in changes in channel geomorphology and physical habitat structure (to be discussed in detail in the next section), including stream bank erosion, stream channel instability, elevated levels of turbidity and fine sediment, channel widening or incision, stream bed scour, and washout of instream structural elements (e.g., large woody debris - LWD).

An extensive study comparing an urban (Kelsey Creek) and a non-urban (Big Bear Creek) stream in the Puget Sound region found that hydrologic changes from urbanization were the principal reasons that the urban stream failed to match its non-urban counterpart in diversity and size of salmonid fish populations and other biological indices (Richey et al. 1981; Perkins 1982; Richey 1982; Scott et al. 1982). The study found that Kelsey Creek had significantly higher stormflows and flood flows, as well as lower baseflows, than Bear Creek. This shift in hydrologic regime resulted in extensive habitat degradation and stream channel alteration from the natural condition.

Another study in the Puget Sound region looked at the streamflow records of six small lowland streams over a 40-year period. Four of the study streams exhibited a significant increase in urbanization and two remained relatively undeveloped over the study period. Each of the urbanized basins experienced a significant increase in flood frequency, while the undeveloped basins showed no discernable corresponding shift. Salmon spawning-count data for the developed basins showed a systematic decline in salmon abundance, while the undeveloped basins showed no evidence of decline. The data imply a link between salmon population decline and either increased flood frequency or an associated degradation in habitat (Moscrip and Montgomery 1997).

The Puget Sound Lowland Stream Research Project (May et al. 1997), one of the most comprehensive studies of the cumulative impacts of urbanization, also found that the shift in hydrologic regime in urbanizing small-stream watersheds was the primary cause of degraded habitat conditions, reduced stream biological integrity, and declining salmon diversity. In the Pacific Northwest, the importance of hydrologic alteration and its effects on stream habitats and the salmonid resource is widely recognized. A significant share of the urban runoff management effort goes into controlling water quantity to attempt to retain pre-development hydrologic patterns. With respect to resource protection, in most other urbanized areas, more attention is generally paid to quality control than to controlling quantity to maintain stream channel integrity. Yet, the same hydrologic modification problems have been noted elsewhere (Wilson 1967; Seaburn 1969; Hammer 1972; Klein 1979).

Finally, a comprehensive literature review conducted by Bunn and Arthington (2002) identifies the key principles and ecological consequences of altered flow regimes resulting from human modification of the watershed. These principles establish the linkages between flow regime and aquatic biodiversity as indicated in Figure 4.

The first principle is that flow is a major determinant of physical habitat in streams, which in turn determines the biotic composition of stream communities. Under this principle, channel geomorphic form, habitat structure, and complexity are determined by prevailing flow conditions. Urban examples of this have been discussed above, including the impact of flashy urban flows on benthic macroinvertebrates and native fish. The biotic communities of streams are largely determined by their natural flow regimes. This is true for aquatic insects and other macroinvertebrates (Resh et al. 1988) as well as fish (Poff and Ward 1989; Poff and Allen 1995; Poff et al. 1997).

The second principle is that aquatic species have evolved life history strategies primarily in direct response to the natural flow regime (Bunn and Arthington 2002). For example, the timing and spatial distribution of salmon migration and spawning in the Pacific Northwest (PNW) is largely determined by the natural flow regimes in each watershed (Groot and Margolis 1991).

The third principle states that the maintenance of natural patterns of longitudinal and lateral connectivity is essential to the long-term viability of many populations of aquatic biota in flowing waters (Bunn and Arthington 2002). Lateral connectivity refers to maintaining a connection between the active stream channel and the floodplain-riparian zone (Ward et al. 1999). This connection is often severely disrupted or lost altogether in urban streams where channelization, and stream bank armoring are common. Longitudinal connectivity is disrupted by fragmentation of the riparian corridor by road or utility crossings (discussed in a later section) and the construction of in-stream migration barriers. The construction of dams and diversion structures, as well as road-crossing culverts that block fish passage, can significantly influence the viability of stream fish populations. Instream barriers can block adult spawning migration, restrict juvenile fish access to rearing or refugia habitat, and disrupt the flow of large woody debris (LWD) and organic matter (OM) within the stream ecosystem. The river continuum concept (Vannote et al. 1980) illustrates the importance of connectivity within a stream ecosystem (Figure 5).

The fourth and final principle states that the survival of invasive, exotic, and introduced (non-native) species is facilitated by altered flow regimes (Bunn and Arthington 2002). The most successful exotic and invasive fish are often those that are either habitat generalists or adaptable to changing conditions (Moyle 1986). Both these strategies are favorable to survival in urbanized hydrologic regimes. In addition, the long-term persistence of invasive fish is much more likely in aquatic systems that are permanently

altered by human activity, as is the case for urbanized watersheds (Moyle and Light 1996).

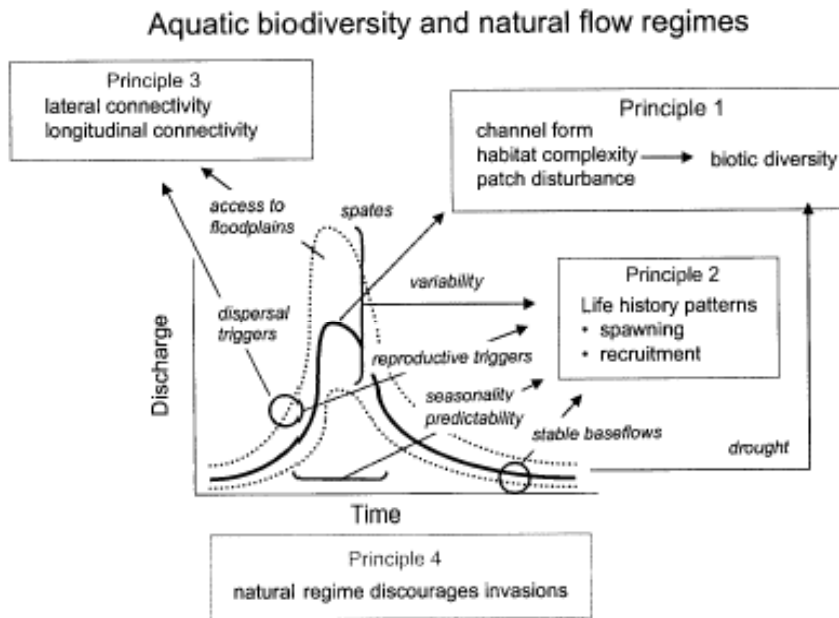


Figure 4 – Relationship between flow regime and ecological integrity (from Bunn and Arthington 2002)

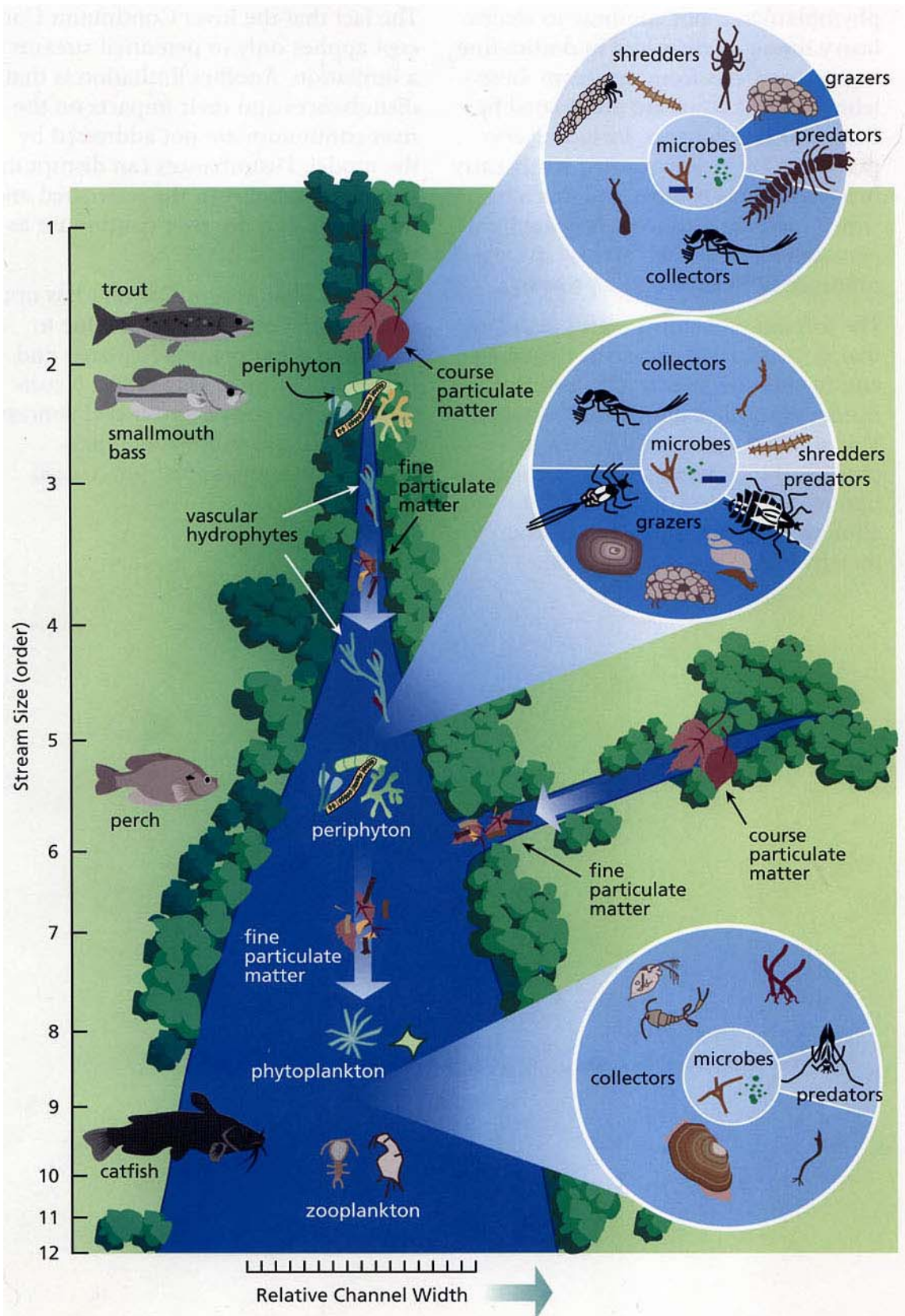


Figure 5. River Continuum Concept (from Vannote et al. 1980).

Hydrological processes and wetlands

Wetlands provide many ecological functions for the watershed in which they are located. These functions include hydrologic, ecological, and water-quality components. Wetlands provide water storage features dispersed throughout the landscape. Riparian wetlands provide natural flood storage volume. Most wetlands also provide critical storage capacity during periods of precipitation. This, in turn, increases groundwater recharge and support of stream baseflows during dry periods. Wetlands also provide key habitat features for a variety of wildlife species. As with streams, watershed development has an impact on wetlands.

The King County Urban Wetland Research Project studied the impacts of urbanization on freshwater wetlands in the Puget Sound lowland ecoregion (Azous and Horner 2001). Water level gauges were used to determine wetland water level fluctuation (WLF). WLF is defined as the difference between base water level (BL) prior to a storm event and the crest, or maximum, water level (CL) for the event ($WLF = CL - BL$). This research found that WLF depends on a variety of watershed and wetland characteristics, but typically exceeded the natural range when basin imperviousness reached 10 percent total impervious area (TIA) (Azous and Horner 2001). Similar results were found in freshwater wetlands in New Jersey (Ehrenfeld et al. 2003) and in tidal wetlands around the country (Thom et al. 2001). In a study in St. Paul, Minnesota, Brown (1988) found that stormwater runoff quantity was related to both the amount of impervious surface area and the wetland-lake area in a basin.

In the Puget Sound urban wetland study, the WLF caused by watershed urbanization was not consistently related to plant species richness but turned out to be an important factor in certain habitat types nonetheless, most notably in emergent wetlands. The frequency and duration of freshwater wetland flooding events was related to plant richness in all Puget Sound wetlands (Azous and Horner 2001). The highest species richness at all water depths was found in wetlands with an average of less than three flooding events per month. Wetlands with a cumulative duration of flooding events lower than three days per month also had the highest species richness (Azous and Horner 2001). While frequency affected plant richness at all water depths, duration particularly compounded the impact of frequency on vegetation found in water over two feet deep. When frequency and duration were analyzed together, it was found that the highest richness was found in wetlands with both an average of less than three events per month and a cumulative duration of flooding that was shorter than six days per month. These two factors were found to be more important than water depth in predicting plant richness (Azous and Horner 2001).

In the Puget Sound lowland ecoregion, watershed urbanization was found to have a negative impact on both native lentic and terrestrial-breeding amphibian richness. Wetlands with increasing urbanization in their contributing watersheds were

significantly more likely to have lower amphibian richness than wetlands in less urbanized or natural watersheds (Azous and Horner 2001). This relationship was linked to increased runoff into urban wetlands as well as a resultant increased WLF. When average WLF exceeded 20 cm, the number of native amphibian species declined significantly (Azous and Horner 2001). It is thought that the greater WLF may have a disproportionate negative impact on amphibian breeding habitat and/or higher egg-embryo mortality due to desiccation of egg masses (Azous and Horner 2001). Urbanized land-use activity in areas immediately adjacent to wetlands (within buffer zones) also decreased native amphibian richness (Azous and Horner 2001). In general, wetlands adjacent to larger areas of forest are more likely to have richer populations of native amphibians.

Wetland WLF and flooding can also affect the richness of bird species. Increased flooding events may inundate nesting sites and disperse pollutants that bioaccumulate in birds through the aquatic food chain (Azous and Horner 2001). Increased runoff and high WLF can alter cover, nesting habitat, and the distribution of birds' food sources. It was not possible, however, to establish that changes in population are directly related to land use since it is difficult to control for all habitat factors besides urbanization. In general, average bird species richness was inversely related to the level of urbanization (Azous and Horner 2001).

The findings of the Puget Sound lowland ecoregion urban wetland study consistently indicated that placing impervious surface on some 10 percent of a watershed creates significantly negative hydrologic, habitat, and ecological responses (Azous and Horner 2001). To complicate the picture, development located immediately adjacent to the wetland (wetland buffer area and surrounding development), rather than away from it, can also have a significant influence on hydrologic conditions, habitat quality, and water quality (Azous and Horner 2001).

Hydrological processes, movement of sediment, and channel structure

Urbanization and the resultant hydrologic changes can cause significant alterations of natural stream morphological characteristics. The direct and indirect impacts of urbanization can affect longitudinal stream channel characteristics such as sinuosity and gradient. In addition, lateral characteristics such as stream channel bankfull width (BFW) and bankfull depth (BFD) can be altered as the stream expands to accommodate the higher runoff-driven flows brought on by watershed urbanization. Figure 6 illustrates the process of channel enlargement in urbanizing streams. Neller (1989) and Booth and Henshaw (2001) both reported that stream channels in urbanized watersheds had cross-sectional areas that were significantly larger than would be predicted based on catchment area and discharge alone.

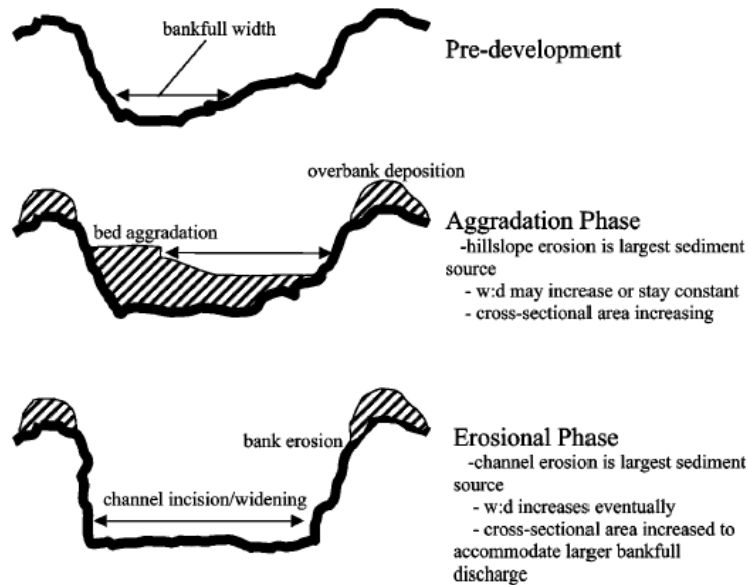


Figure 6 – Urban Stream Morphological Changes (from Booth and Henshaw 2001)

Channel enlargement can be a gradual process that follows the pace of urbanization. Alternatively, it can frequently occur abruptly in response to particular storms (Hammer 1972; Leopold 1973; Booth 1989; Booth and Henshaw 2001). Even in cases where the stream has been stable for many years, sudden and sometimes massive changes in channel dimensions can occur in a single large storm, once urbanization progresses to some critical level. In addition to causing accelerated channel enlargement, the higher and more frequent bankfull flows characteristic of urbanizing streams can also cause stream bank erosion, floodplain degradation, and a loss of channel sinuosity (Arnold et al. 1982).

During the construction phase of development, surface erosion of exposed areas can increase the supply of sediment available to runoff. This deposition of excess sediment can result in streambed aggradation and over-bank deposition in floodplain areas. After construction is complete in a sub-basin, the external supply of sediment is reduced, but bankfull flows continue to increase as runoff from impervious surfaces increases. This can lead to faster stream bank erosion and channel enlargement as the stream tries to accommodate the increased streamflows (Paul and Meyer 2001).

Channel enlargement tends to occur more often in urban streams that have some grade-control structures, such as in-stream LWD or road culverts. In these cases, the stream will generally erode the banks in order to widen the cross-sectional area to carry the higher urbanized flows. Culverts and other artificial grade-control structures can cause downstream scour or upstream sediment deposition if not properly installed and maintained. Culverts in urban streams often become migration barriers for aquatic biota

such as anadromous fish or amphibians. In addition, if not properly sized for urban streamflows, culverts can cause significant localized flooding.

It has been hypothesized that urban streams will eventually adjust to their post-development hydrologic regime and sediment supply. There is evidence that this is the case in some regions, such as Vancouver, British Columbia, Canada (Finkebine et al. 2000) and in the Puget Sound region (Booth and Henshaw 2001) where some urban streams seem to have stabilized several decades after build-out was completed.

In other situations, rapid channel down-cutting, known as incision, can be especially dramatic in urbanizing streams, particularly in regions with unconsolidated soils or where instream (e.g., LWD) structure is lost (Shields et al. 1994). In the Pacific Northwest, incision can result when increased flow and loss of LWD that dissipates energy occur in relatively steep channels with easily erodible substrate (Booth 1991). While all channel damage is ecologically detrimental, incision is especially problematic because it removes virtually all habitat and supplies great quantities of sediment that do further damage downstream (Booth and Henshaw 2001).

Land-use encroachment into floodplain areas and flood control measures such as dikes and levees can also simplify and straighten a stream channel. This can exacerbate fluvial processes causing channel alterations downstream (Graf 1975). In addition to channel modifications carried out during urban development, many streams have residual channelization impacts from past agricultural activities. Stream bank armoring or “rip-rapping” used to mitigate stream bank erosion can actually worsen downstream flooding and stream bank erosion problems. Storm event flows are unable to spread out onto the floodplain, and the increased velocities are transferred downstream along with the elevated sediment loads. There can also be a direct loss of channel migration zone (CMZ) as well as floodplain disconnection, as stream banks are armored and development encroaches. Trimble (1997) demonstrated that channel enlargement, due to the increase in urbanization-driven watershed flows, caused extensive stream bank erosion, which accounted for 66 percent of the sediment transported downstream in an urban stream in San Diego, CA.

Research in several locations suggests that flows larger than a two- to five-year frequency discharge can be sufficient to create large-scale channel disruption (Carling 1988; Sidle 1988; Booth 1990). More than anything else, the greatly increased incidence of these flows explains the ecological vulnerability of urban streams. In addition to stream bank erosion and streambed scour or incision, higher urban streamflows can physically destroy or wash out instream structural elements, such as LWD. This can have a negative feedback effect on the stream channel. As higher flows wash out more and more LWD, the channel becomes even more unstable and more susceptible to further geomorphic degradation. Under these conditions, the stability of stream channels can actually unravel as the combined effects of channel incision, enlargement, and erosion

continue to impact the stream system (Horner et al. 1997).

Two similar studies, one in Maine (Morse 2001) and one in the Puget Sound region (May et al. 1997), demonstrated that stream bank erosion was related to the level of watershed imperviousness and linked directly to the shift in hydrologic regime. This is not to say that stream bank erosion and other geomorphic changes are only driven by urbanization. Booth (1991) and Bledsoe (2001) both reported that geomorphic change in response to urbanization depends on other factors, such as underlying geology, vegetation structure, and soil type.

Stream bank erosion and streambed scour resulting from the altered urban flow regime described previously can result in the production of excessive quantities of fine sediment (Nelson and Booth 2002). This increase in sediment yield can be especially acute during the construction phase of development when runoff from bare ground on construction sites can carry very high sediment loads. The alteration in sediment transport regime can change stream from a single, meandering channel to a braided and aggrading form (Arnold et al. 1982).

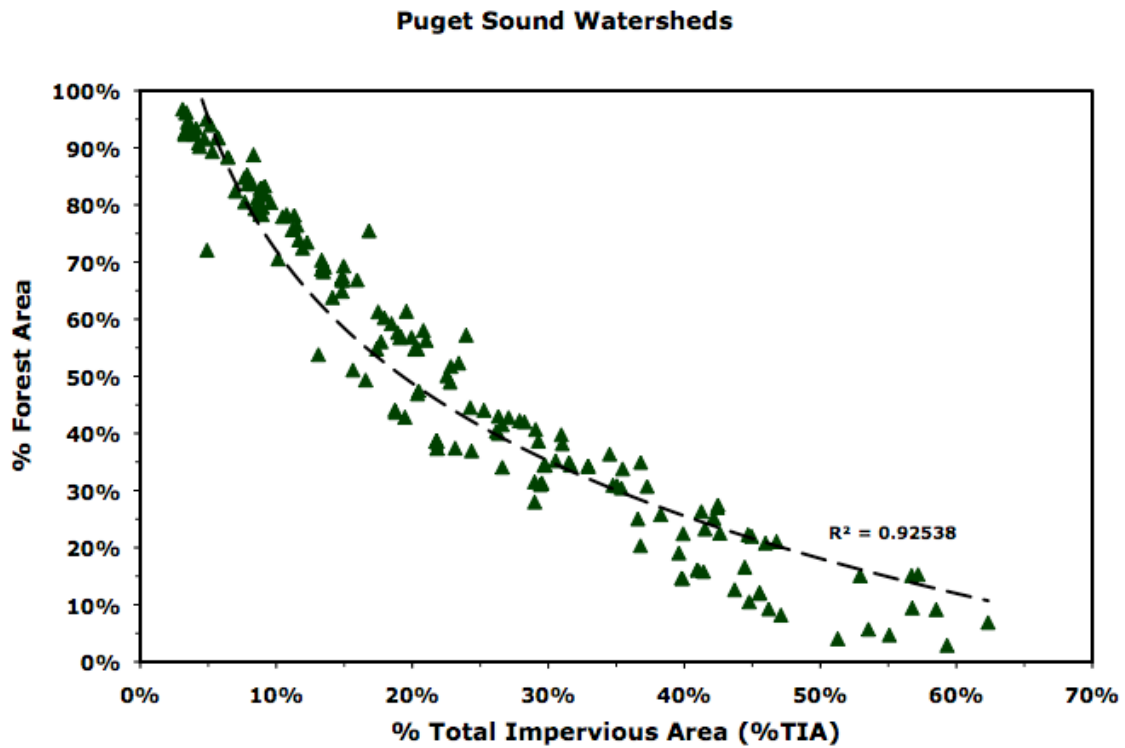
The shift in sediment transport regime that typically accompanies urbanization can also result in excessive sedimentation of streambed habitats. Streambeds can also become embedded and ecologically non-functional with frequent deposits of fine sediment. In the Puget Sound region, it was found that the percentage of fine sediment occurring in stream substrates used by salmon for spawning increased along with watershed urbanization (May et al. 1997).

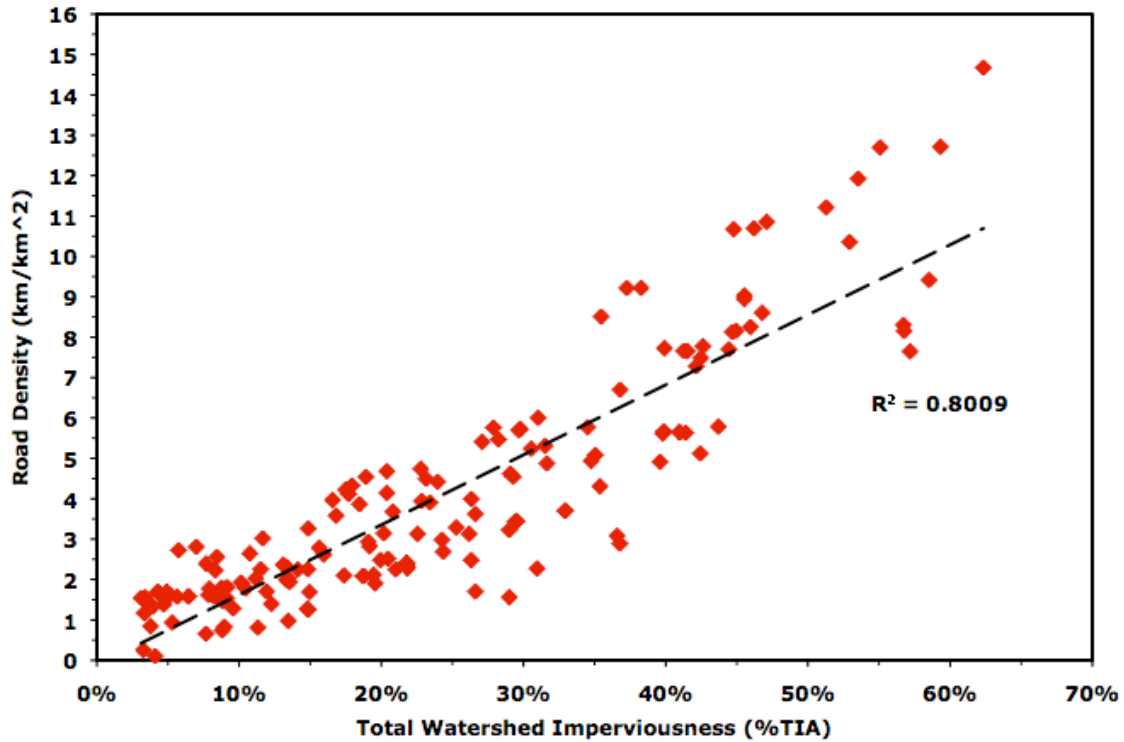
When a watershed is finally fully built out, this situation can actually reverse as impervious surfaces become the dominant landscape feature. Under fully urbanized basin conditions, there is often a lack of sediment delivered to stream channels (Wolman 1967; Booth 1991; Pizzuto et al. 2000). Under highly urbanized conditions, streambeds can become effectively armored and are, for the most part, ecologically non-functional (May et al. 1997).

As discussed above, the geomorphologic impacts of watershed urbanization include the following:

- Stream channel enlargement and instability
- Stream bank erosion and fine sediment production
- Stream channel incision or down-cutting
- Streambed scour and fine sediment deposition
- Increase in streambed embeddedness
- Riparian buffer (lateral) encroachment
- Riparian corridor (longitudinal) fragmentation
- Channelization and floodplain encroachment

- Stream bank armoring and loss of the channel migration zone (CMZ)
- Increased sediment yields, especially during construction
- Washout of instream LWD
- Simplification of the natural drainage network, including loss of headwater channels and wetlands and lower drainage density
- Modification of natural instream pool-riffle structure
- Fish and amphibian migration barriers (e.g., culverts and dams)





Water quality: delivery and routing of nutrients and toxicants

In addition to the hydrologic and physical impacts of stormwater runoff generated by the urbanization process, there are water-quality impacts to aquatic ecosystems and biota that result from exposure to the pollutants found in urban runoff. Stormwater runoff from urbanized areas is generated from a number of sources, including residential areas, commercial and industrial areas, roads, highways and bridges. Essentially, as discussed earlier, any surface that does not have the capability to store and infiltrate water will produce runoff during storm events. These are the previously-discussed impervious surfaces. As the level of imperviousness increases in a watershed, more rainfall is converted to runoff.

Impervious surfaces (roads, parking lots, rooftops, etc.) are the primary source areas for pollutants to collect within the built environment. Runoff from storm events then carries these pollutants into natural waters via the stormwater conveyance network. The land use (e.g., residential, commercial, and industrial) and human activities (e.g., industrial operations, residential lawn care, and vehicle maintenance) characteristic of a drainage basin largely determine the mixture and level of pollutants found in stormwater (Weibel et al. 1964; Griffin et al. 1980; Makepeace et al. 1995; Pitt et al. 1995).

Stormwater is a form of non-point source (NPS) pollution and typically contains a mixture of contaminants, including metals, petroleum hydrocarbons, and organic

toxicants (i.e., pesticides, herbicides, and industrial chemicals). The National Urban Runoff Program (NURP) identified stormwater as a significant source of potentially toxic pollutants to receiving waters (EPA 1983). Other studies have confirmed the NURP findings and improved the level of knowledge with regard to stormwater pollution impacts (Ragan and Dietermann 1975; Pitt and Bozeman 1982; Field and Pitt 1990; Bannerman et al. 1993). Two of the most common stormwater pollutant components are petroleum hydrocarbon compounds and metals (e.g., zinc, copper, lead, etc.). Hydrocarbon sources include vehicle fuels and lubricants (Hoffman et al. 1984; Fram et al. 1987; Smith et al. 1997). Metals are also associated with vehicle maintenance, roads, and parking areas (Wilber and Hunter 1977; Davies 1986; Field and Pitt 1990; Pitt et al. 1995). Pesticides, herbicides, and other organic pollutants are commonly found in stormwater flowing from residential and agricultural areas (Pereira et al. 1996; Black et al. 2000; Foster et al. 2000; Hoffman et al. 2000). Studies in Puget Sound confirm these findings (Hall and Anderson 1986; May et al. 1997; Black et al. 2000). In many cases, even banned pesticides, such as DDT or other organochlorine pesticides, can be found in urban stream sediments. Toxic industrial compounds such as PCBs can also be present in urban runoff (Black et al. 2000). In general, the more intense the level of urbanization, the higher the pollutant loading. Also, the greater the diversity of land use activities, the more diverse the mixture of pollutants found in stormwater runoff (Herricks 1995; Makepeace et al. 1995; Pitt et al. 1995).

The transport and fate mechanisms of stormwater pollutants in receiving waters tend to be highly variable and site-specific. Pollutants are often transported from source areas (roads, parking lots, lawns, etc.) to receiving waters via roadside ditches, stormwater pipes, or by atmospheric deposition. In general, the concentration of pollutants found in stormwater runoff is much higher than that found in receiving waters, due mostly to dilution and removal mechanisms. In addition, most stormwater pollutants are typically found in particulate form, attached to fine sediment particles and organic matter (Pitt et al. 1995). This is especially true for nutrients, organics, and metals. In most cases, the particulate forms of toxic pollutants tend to be less bio-available (Herricks 1995).

Because of the potential for accumulation of pollutants in sediment and the potential of sediments as sources of toxics, contaminated sediments likely play an important role in many of the biological impacts associated with stormwater runoff. In general, most pollutants, especially metals, are found in particulate forms within the water column or sediments, and pollutant concentrations tend to be higher for smaller sediment particle sizes (DePinto et al. 1980).

Physical variables such as flow regime and instream habitat are important to native biota, as are chemical factors like water or sediment quality. Human activities in urbanizing watersheds can lead to both physico-chemical pollution and biophysical alterations of stream habitats. The evaluation of cumulative ecological impacts in urban areas can be problematic where both types of stressors occur. The relative importance of

one stressor as compared to another is difficult to quantify, especially when antagonistic or synergistic effects are present. For example, effects of contaminants can also be masked by instream or riparian habitat degradation. All of these variables need to be quantified in order for a complete assessment of the impact of stormwater on human health, aquatic ecosystems, and instream biota to be developed (Horner et al. 1997).

Current stormwater monitoring and impact assessment programs indicate that the most likely cause for degradation of biological integrity in receiving waters is a combination of physical habitat degradation, changes in the hydrologic regime, food web disruptions, and long-term exposure to anthropogenic contaminants (Pitt 2002). However, chronic or acute exposure to potentially toxic contaminants may be especially problematic for benthic organisms such as macroinvertebrates and for organisms that have a benthic life stage (e.g., salmonids during their embryonic development stage). Acute toxicity of aquatic biota due to exposure to stormwater runoff in receiving waters is rare (Pitt 2002).

Current research appears to indicate that even when stormwater toxicity is high, it is only for short periods of time during episodic storm events. It has been hypothesized that relatively short periods of exposure to toxic compounds at the levels normally found in stormwater are not sufficient to produce mortality in aquatic organisms. This is often based on the assumption that most of the toxic chemicals found in stormwater are found in particulate form and are not bio-available. This school of thought holds that most of the toxicity problems observed in urban receiving waters are a result of illegal discharges or dumping and that the risk from stormwater and sediment-bound toxics is low. However, this view tends to ignore the cumulative impacts of frequent exposures of organisms in receiving waters to stormwater as well as the potential release of toxics from sediments due to changes in ambient water chemistry. In reality, urban stormwater runoff has been found to cause significant impacts on aquatic biota in receiving waters (Burton and Pitt 2001).

Evaluation of stormwater or receiving water quality is a complex and expensive project. The type and quantity of stormwater constituents are highly variable, depending on land use and human activities in the source area of concern. There are also numerous confounding factors that influence how stormwater interacts with receiving waters. In addition, the relationship between observed biological effects on receiving water and possible causes (including stormwater-related toxicity) are especially difficult to identify, let alone quantify. Countless antagonistic and synergistic chemical relationships exist among the constituents in stormwater runoff and receiving waters. Physico-chemical transformations can render toxic substances harmless or create toxic mixtures from individually harmless compounds. Contaminants can also be associated with suspended sediment particles or mobilized from streambed sediments due to scour during high-flow events (Mancini and Plummer 1986). It is likely that in most situations, multiple stressors and cumulative impacts play a significant role in the decline of biological integrity.

Many studies have shown the detrimental effects of stormwater runoff on receiving water biota. However, few studies have demonstrated a direct cause-and-effect relationship between stormwater and toxicity to aquatic biota. Beginning with the National Urban Runoff Program (NURP) (EPA 1983), numerous studies have focused on determining the chemical characteristics of stormwater. An update of the NURP stormwater data was conducted in 1999 (Smullen et al. 1999). There have also been several studies on the toxicological effects of stormwater on aquatic biota. Pitt and Bozeman (1982) studied the impacts of urban runoff on stream water quality and biological conditions in Coyote Creek in the San Francisco Bay area. The results of this study indicated that water and sediment quality were significantly degraded by urban stormwater runoff (Pitt and Bozeman 1982). There was also some evidence of bioaccumulation of urban pollutants in plants, fish, and macroinvertebrates resident in the system (Pitt and Bozeman 1982).

Studies of urban streams in Bellevue, Washington examined the ecological and biological impacts of stormwater runoff (Perkins 1982; Richey 1982; Scott et al. 1982; Pitt and Bissonette 1984). These studies documented physico-chemical water quality and instream habitat degradation due to watershed development and stormwater runoff. Massive fish kills in Kelsey Creek were also observed during one of these studies. These fish kills were attributed to illegal dumping of toxic chemicals into local storm drains.

Medeiros and Coler (1984) used a combination of laboratory flow-through bioassay tests and field experiments to investigate the effects of urban stormwater runoff on fathead minnows. They observed chronic effects of stormwater toxicity on growth rates in the test organisms.

Hall and Anderson (1988) studied the effects of urban land use on the chemical composition of stormwater and its toxicity to aquatic invertebrates in the Brunette River in British Columbia. This study found that land use characteristics and the antecedent dry period between rainfall events had the greatest influence on stormwater quality and toxicity. Toxicity magnitude study decreased across the land use sequence: commercial > industrial > residential > open space (Hall and Anderson 1988). This study also identified the "first flush" effect as being significant from a toxicity standpoint. The longer the dry build-up period between storms, the higher was the pollutant load and the greater the toxicity of stormwater runoff (Hall and Anderson 1988).

A study of stormwater toxicity in Birmingham, Alabama used toxicity screening as the primary detection method (Pitt et al. 1995). Of the source area samples collected, 9 percent were classified as extremely toxic, 32 percent were moderately toxic, and 59 percent showed no evidence of toxicity. Vehicle service and parking areas had the highest levels of pollutants and potential toxicants. Metals and organics were the most common contaminants found in stormwater samples.

A field study in Milwaukee, Wisconsin investigated the effects of stormwater on Lincoln Creek (Crunkilton et al. 1997). Streamside toxicity testing was conducted using flow-through aquaria with fathead minnows. In addition, instream biological assessments were conducted along with water and sediment quality measurements. The results of the flow-through tests showed no toxicity in the fathead minnows until 14 days after exposure and 80 percent mortality after 25 days of exposure, indicating that short-term toxicity testing likely underestimates the true toxicological impact of stormwater in receiving waters.

A study in North Carolina found that stormwater runoff from vehicle service and fueling stations had consistently elevated levels of polyaromatic hydrocarbon (PAH) compounds, methyl tertiary butyl ether (MTBE), and other potentially toxic contaminants (Borden et al. 2002).

Runoff from agricultural or landscaped areas can also contain significant levels of potential toxicants, especially pesticides and herbicides (Liess et al. 1999; Thomas et al. 2001; Neumann et al. 2002; Arnold et al. 2004). These toxicants are also common in stormwater runoff from residential and urban landscaped areas (Pitt et al. 1995).

Sediment contaminated by stormwater runoff also has a detrimental effect on receiving water biota. Many of the observed biological effects associated with stormwater runoff and urban receiving waters may be caused by contaminated sediments, especially those impacts observed on benthic organisms. In addition, mortality of benthic invertebrates can be high in urban streams, especially during low flow periods, suggesting that toxicity associated with exposure to contaminated sediment, concentration of toxics in the water column, and/or ingestion of contaminated organic particulates is to blame (Pratt et al. 1981; Medeiros et al. 1983; Black et al. 2000).

Studies of urban stream sediments have shown the effects of metal toxicity on early life stages of fish and invertebrates (Boxall and Maltby 1995; Hatch and Burton 1999; Lieb and Carline 2000). Developmental problems and toxicity have been attributed to the contaminant accumulation in sediments and the remobilization of contaminated sediments during storm events. Hatch and Burton (1999) also observed significant toxicity at a stormwater outfall site, where sediments were found to be contaminated by multiple stormwater-related pollutants. Lieb and Carline (2000) showed that metals were more prevalent in sediments downstream of a stormwater treatment pond than upstream in a natural area. However, no acute toxic effects were noted. Zinc (Rose and Peters 2001) and copper (Boulanger and Nikolaidis 2003) are the most common metals found in urban sediments contaminated by stormwater runoff. These metals can be quite mobile under typical conditions found in urban receiving waters, but in most cases, a majority of the metal ions are bound to fine sediment particles and are not generally bio-available. Examples of elevated levels of stormwater-related toxicants accumulating in urban stream sediments are numerous (Pitt 2002). The levels of metals

in urban stream sediments are typically orders of magnitude greater than those in the water column (DePinto et al. 1980; Pitt and Bozeman 1982; Scott et al. 1982; May et al. 1997). Similar results are found when analyzing marine sediments from urban estuaries with stormwater discharges (Long et al. 1995; Morresey et al. 1997; Bolton et al. 2003).

Water quality and wetlands

In a study of Puget Sound Basin freshwater wetlands (Azous and Horner 2001), many water quality parameters exhibited upward trends with increased urbanization. Median pH levels were particularly elevated in highly urbanized wetlands while dissolved oxygen (DO) levels experienced more modest increases. Median conductivity and NH₃ levels were also significantly higher in urbanized wetlands than in non-urbanized wetlands. Finally, similar rates of increase in median concentrations of total suspended solids (TSS), soluble reactive phosphorus (SRP), fecal coliforms (FC), lead (Pb) and zinc (Zn) were found with each step in the urbanization process (Azous and Horner 2001).

In the wetlands studied, low concentrations predominated, indicating minimal water quality impacts. Concentrations of lead (Pb), however, tended to violate water quality criteria for the protection of aquatic life (Azous and Horner 2001). In both urbanized and non-urbanized wetlands, wetland morphology type was associated with varying levels of water quality parameters. Morphology refers to the shape, perimeter length, internal horizontal dimensions, and topography of the wetland as well as to water pooling and flow patterns. Higher levels of DO, pH, conductivity, NO₃+NO₂-N, SRP, FC, and Pb were found in flow-through wetlands. Flow-through wetlands (FT) are channelized and have clear flow gradients, while open water wetlands (OW) contain significant pooled areas with little or no flow gradient (Azous and Horner 2001). A large proportion of FT wetlands is found in urban areas, due to wetland filling, stream channelization, and higher peak runoff flows. This may help explain why pollutant level trends are higher in these wetlands (Azous and Horner 2001).

In the Puget Sound wetlands study, sediment samples exhibited similar trends in urban and flow-through wetlands as the water quality parameters discussed previously. Median pH levels increased with each successive level of urbanization (Azous and Horner 2001). Metals, including Pb, Zn, As (arsenic) and Cu (copper) also generally tended to increase with urbanization. As with water quality samples, sediment metal concentrations did not exceed severe effect thresholds based on the Washington State Department of Ecology standards. Some Cu and Pb mean and median concentrations exceeded lowest effect thresholds (Azous and Horner 2001). While these metals tended to be found in greater concentrations in urban wetlands, they can also be found at elevated levels in non-urban wetlands. High Cu, Pb and TPH levels were seen in the two most impacted urban wetlands (Azous and Horner 2001). Thus, local conditions may be more important factors in determining soil metal concentrations. These could include the delivery of metals via precipitation, atmospheric dry-fall, dumping of metal trash,

and leaching from old constructed embankments (Azous and Horner 2001).

The impact of human activity and development on water quality varies widely between wetlands of different urbanization levels. For moderately urbanized wetlands, there is a mixed picture. Median total dissolved nitrogen concentrations (ammonia, nitrate, and nitrite) have been found to be more than 20 times higher than dissolved phosphorus, but phosphorus is the most important factor limiting plant and algal growth. As would be expected, these wetlands exhibit slightly elevated pH levels (median pH = 6.7). Dissolved oxygen is well below saturation, at times below 4 mg/l. Dissolved substances tend to be higher than in non-urbanized wetlands but are also somewhat variable. Suspended solids are only marginally higher than in non-urbanized wetlands but are also variable (Azous and Horner 2001).

In highly urbanized wetlands, water quality samples revealed higher nutrient levels. Unlike non-urbanized or even low-to-moderately urbanized wetlands, these are likely to have median NO₃ + NO₂-N concentrations above 100 mg/l and total phosphorus (TP) over 50mg/l (Azous and Horner 2001). In one study, FC and EC were shown to be significantly higher in highly urbanized wetlands. Many of these wetlands were within watersheds with low-density residential development; bacterial contamination was experienced even at suburban levels of development (Azous and Horner 2001).

An effort was made to correlate water quality conditions with watershed and wetland morphological characteristics. Acidity (pH), TSS, and conductivity showed the strongest ability to predict watershed and morphology characteristics. Pollutants such as TP, Zn and FC, which are often absorbed to particulates, also exhibited strong correlations with watershed conditions and morphology (Azous and Horner 2001). On the other hand, forest cover was the best predictor of these water quality parameters. The next-best land cover predictors of water quality were the percentage of impervious surface, forest-to-wetland areal ratio and morphology (Azous and Horner 2001). A rise in the total impervious area will facilitate the delivery of TSS and increase conductivity. TSS and conductivity are directly and indirectly harmful to wetland biological communities (Azous and Horner 2001).

These results suggest that water quality impacts can be minimal for a range of deforestation and development levels, but will become degraded beyond some threshold. Effective impervious area, which is the amount of land drained by a storm drainage system, was more predictive of water quality than total impervious area. As total impervious area approaches a range of 4 to 20 percent and forested area declines to between 0 to 15 percent, water quality will begin to decline (Azous and Horner 2001).

Wetlands in developing areas are especially vulnerable to erosion caused by construction, which contributes to increased sediment levels. During these periods, both mean and median TSS values increase, although mean values show the greatest change. After construction is completed, and more surface area is covered with structures and

vegetation, these values return to their approximate values before development. The sediments contributed by this erosion carry pollutants such as phosphorus and nitrogen (Azous and Horner 2001).

Development also affects soils in wetlands. In Puget Sound Basin wetlands, somewhat elevated pH levels prevailed. These soils were often aerobic, although many times their redox potentials were below levels at which oxygen is depleted. Metals such as Cu, Pb and Zn were higher in developing areas but did not usually approach severe effects thresholds (Azous and Horner 2001). In a synoptic study of 73 wetlands, about 60 percent of which were urban and the rest non-urban, Pb levels were significantly different in both the inlet and emergent zones (Azous and Horner 2001). In some soil samples, high toxicity levels were probably caused by the extraction and concentration of naturally occurring organic soil compounds during laboratory analysis. Samples from two wetlands, however, probably contained anthropogenic toxicants because the results indicated toxicity in the absence of any visible organic material (Azous and Horner 2001).

For each region studied in the Puget Sound area, a regression was developed between the presence of crustal metals and toxic metals in relatively unimpacted wetlands. If the concentration of a toxic metal was above a 95 percent confidence level, it was probable that the metals were of anthropogenic origin. The results of this analysis echoed those described previously for urbanized wetlands. The results revealed greater toxic metal enrichment in the most urban wetlands (Azous and Horner 2001).

Water quality and estuarine waters

The effects of watershed development and stormwater runoff extend into marine waters at the mouths of streams (sub-estuaries) and in the nearshore environment of coastal regions. As with freshwater receiving waters, these impacts include physical, chemical, and biological effects. Several studies on the toxic effects of water pollution on salmon have been conducted in the Puget Sound region and the Lower Columbia River Estuary (McCain et al. 1990; Varanasi et al. 1993; Casillas et al. 1995a, 1995b; Casillas et al. 1998a, 1998b; Collier et al. 1998). In these studies, there were demonstrable chronic toxicological effects (immuno-suppression, reduced disease resistance, and reduced growth) of PAHs, PCBs, and other organic pollutants seen in juvenile and adult salmon.

In a study of multiple stormwater discharge sites in Massachusetts Bay, high levels of PAH compounds were found in receiving waters and estuarine sediments (Menzie et al. 2002). Land use was a critical factor in determining pollutant composition and concentrations, with urbanized areas (mixed residential, commercial, and industrial land uses) having the highest pollutant (PAH) levels.

A study of the Hillsborough River in Tampa Bay, Florida investigated the impacts of

stormwater on estuarine biota. Plants, animals, sediment, and water quality were all studied in the field and supplemented by laboratory bioassay tests. No significant stormwater toxicity-related impacts were noted (MML 1984).

A study of stormwater discharges from Chollas Creek into San Diego Bay, California, indicated measurable toxic effects to aquatic life (Schiff et al. 2003). This study found that a toxic plume from the freshwater creek extended into the estuary, with the highest toxicity observed closest to the creek mouth. The toxicity decreased with increasing distance from the mouth due to mixing and dilution. Toxicity identification evaluation (TIE) methods were used, and it was found that trace metals from stormwater runoff were most likely responsible for the plume's toxicity to the sea urchins used in this study (Schiff et al. 2003).

A study of the water quality impacts of stormwater runoff into Santa Monica Bay, California also identified toxic effects in the estuarine receiving waters (Bay et al. 2003). As in the San Diego study, the freshwater plume from an urbanized stream (Ballona Creek) was responsible for the toxicity observed in marine organisms. Stormwater-transported metals (mainly zinc) were identified as the most likely toxic constituent. The only toxic effects noted were chronic, not acute. As in the previously discussed study, the toxicity decreased with increasing distance from the mouth due to mixing and dilution (Bay et al. 2003). Sediments in estuarine areas were also highly contaminated by stormwater pollutants (Schiff and Bay 2003).

Water quality impacts on aquatic biota

Several studies on the toxic effects of stormwater runoff on native biota have been conducted in the Puget Sound region. One of the first studies looked at the uptake of aromatic and chlorinated hydrocarbons by juvenile chinook (McCain et al. 1990). This study found no acute toxicity, but identified numerous potential chronic impacts on growth and survival. In a related study, juvenile chinook salmon from both a contaminated urban estuary and a non-urban estuary were studied for two years (Stein et al. 1995). Exposure to aromatic and chlorinated hydrocarbons was measured, and both PAH and PCB levels in fish from the urban estuary were significantly higher than in fish from the non-urban estuary. The results of these studies indicate that out-migrant juvenile salmon have an increased exposure to chemical contamination in urban estuaries during their residence time in these habitats. This exposure was determined to be sufficient to elicit biochemical responses and to have the potential for chronic toxicity effects (Stein et al. 1995).

Runoff from urban areas can also contain significant levels of pesticides and herbicides that have been shown to be potentially toxic to native biota (Bortleson and Davis 1997; MacCoy and Black 1998; Voss et al. 1999; Black et al. 2000; Hoffman et al. 2000). In a study conducted by King County, Washington, pesticides and herbicides in runoff and

urban streams were linked to retail sales of the same pesticides within the urban watersheds under study (Voss and Embrey 2000). The most common pesticides and herbicides detected during storm events included diazinon, 2-4-D, dichlorbenil, MCP, prometon, and trichlopyr (Voss and Embrey 2000).

Diazinon has been shown to have neurotoxic effects on salmon (Scholz et al. 2000). At sublethal levels, it was shown to disrupt homing behavior in chinook salmon by inhibiting olfactory-mediated responses (Scholz et al. 2000). This may have significant negative consequences for the survival and reproductive success of native salmonids.

Short-term exposures to copper (such as during storm runoff events in urban areas) have also been demonstrated to have sublethal effects on coho salmon by inhibiting the olfactory nervous system (Baldwin et al. 2003). In this study, the neurotoxic effects of copper were found to be dose-dependent, having a measurable impact over a broad range of concentrations. These effects occurred rapidly upon exposure. The study concluded that short-term exposures can interfere with olfactory-mediated behaviors in juvenile coho salmon and may impact survival or migratory success of native salmonids (Baldwin et al. 2003).

Metals are a significant pollution component of urban stormwater runoff and non-point source (NPS) pollution. Heavy metals are of particular interest because many cannot be chemically transformed or destroyed and are therefore a potential long-term source of toxicity in the aquatic environment. Although the specific metals and their concentrations may vary widely depending on the anthropogenic sources present, they are common to almost all water pollution. Many trace metals are important as micronutrients for both plants and animals, playing essential roles in metabolism and growth. These include iron (Fe), zinc (Zn), copper (Cu), and Manganese (Mn), to name a few. Nutrient requirements vary between species, life stages, and sexes, but normal concentrations of these micronutrient trace metals are low and typically fall within a narrow acceptable band. Exposure to concentrations outside the optimal range can have deleterious or even toxic effects. Other trace metals, which are not essential, such as lead (Pb), cadmium (Cd), and mercury (Hg) can be toxic at very low levels, either acutely or due to chronic/long-term exposure. Aluminum (Al), chromium (Cr), and nickel (Ni) are also found in urban runoff.

Anthropogenic sources of metal pollution are common throughout the environment. These include industrial processes, mining, and urban storm runoff. Urban runoff can contain a wide variety of trace metals from sewage discharges, fossil-fuel combustion, automobile traffic, anti-corrosion products, and various industrial sources. In general, the concentration, storage, and transport of metals in urban runoff or streams are closely related to organic matter content and other sediment characteristics. Fine sediment, especially the organic fraction, has a high surface area and a high binding capacity for metals, resulting, as mentioned above, in generally higher levels of metal contamination

in sediments than in the water column (Rhoads and Cahill 1999).

Several studies have been conducted to characterize the levels of metals in stormwater runoff, receiving waters, and sediments (Bryan 1974; Wilber and Hunter 1977; Pitt et al. 1995; May et al. 1997; Neal et al. 1997; Sansalone and Buchberger 1997; Barrett et al. 1998; Wu et al. 1996; Wu et al. 1998). Generally, the levels of various metals in stormwater are quite variable and dependent on a number of factors, including background watershed characteristics, land use practices, and specific sources.

Certain urban stream organisms, including algae, arthropods, mollusks, and annelids, have exhibited elevated levels of metal concentrations (Davis and George, 1987). Ecological responses to metals occur at all levels in the ecosystem and include the loss of sensitive taxa, both chronic and acute toxicity effects, and altered community structure. One study (Pitt et al. 1995) of urban stormwater samples, using the Micro-Tox toxicity-screening procedure, found that less than 10 percent of samples were classified as extremely toxic, a bit over 30 percent were moderately toxic, and the majority (about 60 percent) showed no evidence of toxic effects. The Micro-Tox methodology was only used to compare relative toxicities of various samples and not as a measure of absolute toxicity or to predict long-term toxic effects of stormwater on receiving waters. Therefore, typically in all but a few heavily polluted systems, the level of toxicants in urban runoff is typically near detection limits (Pitt et al. 1995).

The toxicity of metals to aquatic plants and organisms is influenced by chemical, physical, and biological factors. Water chemistry characteristics such as temperature, pH, alkalinity, and hardness all affect metal toxicity. Physical aspects of exposure, such as metal speciation, duration of exposure, intensity of exposure events, and inorganic or organic ligand binding, also have a significant bearing on metal toxicity (Davies 1986). Bioavailability of metals, the life stage of the affected organisms, organism health, and the natural sensitivity of the species involved are also important determinants of metal toxicity. Aquatic toxicology data generally indicates that the ionic fraction of metals constitutes the primary toxic form (Roline 1988).

Acute toxicity to aquatic organisms can be manifested as a wide range of effects, from reduced growth rate to mortality. Laboratory studies on the mechanism of toxicity of zinc to fish in general indicate that zinc causes death via gill hypoxia (excess mucous secretion and suffocation) and gill tissue necrosis (Davies 1986). Osmoregulatory failure appears to be the most likely effect of acute copper toxicity. Lead and mercury affect the central nervous system coordination of activity in fish, as well as interfering with cellular osmoregulation (Pagenkopf 1983). The metal species present in solution and the ambient water chemistry can have a significant influence on metal toxicity. Consideration of total metal concentration alone can be misleading because chemical speciation of trace metals significantly affects the bioavailability to aquatic organisms and thus the ultimate toxicity (Davies 1986). For the most part, organisms assimilate

uncomplexed metal ions more readily than complexed forms. Increases in pH, alkalinity, and hardness generally decrease metal toxicity. Hardness (Ca^+ and Mg^{++}) has an antagonistic effect on metal toxicity in that the calcium and magnesium ions compete with metal ions for uptake sites on the gill surfaces, thus reducing the toxic effects of the metal ions (Davies 1986). Alkalinity reduces metal toxicity through the buffering mechanism of the carbonate system. Under pH control, the carbonate and bicarbonate ions complex metal ions into soluble or insoluble, less toxic forms (Pagenkopf 1983). In most cases, in alkaline waters, metals do not reach toxic levels until their concentration overwhelms the natural buffering capacity of the carbonate system. Organic ligands can also complex metal ions, thus reducing toxicity by binding metals to particulates and making them relatively non-bioavailable. Metal toxicity generally increases when ambient temperature rises, due to the combined effects of an increase in both organism metabolism and chemical activity. Light intensity may also have a synergistic effect on the toxicity of some metals.

Chronic toxicity of metals is generally most apparent in the embryonic and larval stages of aquatic organisms and the early life stages of aquatic plants. As a period of rapid development, the early life stage is the most sensitive stage of the organism's life cycle for metal toxicity in general and other toxicants as well. Embryogenesis is a particularly sensitive period for fish with regard to metals (Davies 1986). The period of larval settlement is the critical phase in invertebrate life history, although invertebrates as a whole are generally less sensitive than fish to trace/heavy metal toxicity (Nehring 1976; Winner et al. 1980; Pratt et al. 1981; Garie and McIntosh 1986). Chronic and sub-lethal effects of metals include reduced growth rates, developmental or behavioral abnormalities, reproductive effects, interference with metabolic enzyme systems, anemia, neurological defects, and kidney dysfunction (Davies 1986). Due to the greater sensitivity of young organisms to metals, any exposures during embryonic development or rearing periods can, apart from the immediate effects, also manifest themselves in the adult organisms. There has been some indication that fish exposure to very low levels of metals during early life stages can result in an acclimation effect, making them somewhat more resistant to future periodic exposures (Davies 1986). As with most toxicants, metal toxicity also increases with exposure period. Therefore, the intermittent nature of urban runoff may be less harmful to some aquatic life forms than continuous exposure to elevated metal concentrations. Bioaccumulation of metals in organisms is also highly variable, depending on the particular metal, its chemical form, the mode of uptake, and the storage mechanisms of the organism. In low alkalinity (soft) waters, most metal species are of the "free" form. In alkaline (hard) waters, more metal ions are complexed, but some portion may remain in the ionic forms, especially if the buffering capacity of the natural water is overwhelmed. System pH also plays a major role in determining the speciation of the metal forms in freshwater (Davies 1986). The rate of chemical (metals) reactions or chemical kinetics is also important to understanding the overall metal toxicity process. Such reactions as complexation do not occur instantaneously in natural waters. In the case of stormwater, runoff time scales may not

allow sufficient time for complexation to take place, thus mitigating or negating the toxicity-reducing buffering effects (Pitt et al. 1995).

The use of aquatic insects and other macroinvertebrates as indicators of the biological integrity of lotic ecosystems is not new. One of the earliest field studies (Nehring 1976) involved using aquatic insects as biological monitors of heavy metal pollution in the analysis and prevention of fish kills. Macroinvertebrates are generally more tolerant of metal pollution than most species of fish found in western streams (e.g. salmonids, sculpins, etc.) and tend to bioaccumulate metals in proportion to the in-water concentration (Nehring 1976). In contrast to the more mobile fish species, macroinvertebrates are relatively sessile organisms. They also constitute an important part of the lotic food web, being the primary food source of most stream fishes. This makes them a useful surrogate for the economically and culturally important fish that inhabit the streams of the western states. In addition, some species of macroinvertebrates turned out to be more sensitive to metal pollution than others. This concept of "tolerant" and "sensitive" groups/species has become an important aspect of macroinvertebrate-based indices of pollution (Winner et al. 1980). In general, stoneflies (Plecoptera) and mayflies (Ephemeroptera) are sensitive to metal pollution, caddisflies (Trichoptera) are moderately sensitive/tolerant, and midges (Chironomids) are metal pollution-tolerant (Garie and McIntosh 1986).

Field studies into the impact of urban runoff on lotic systems often use macroinvertebrate community structure as an indicator of ecosystem degradation. Many studies have found that, although urban runoff is the causal agent of ecosystem disruption, the impacts of stormwater pollution events are not just short-term. Partitioning of pollutants, especially metals, into sediments has been shown to have long-term ecological consequences on the primarily benthic-dwelling macroinvertebrate community structure (Pratt et al, 1981). In many cases, analysis of stormwater samples will not detect significant metals either in the dissolved or particulate form, but sediment samples will show metal accumulation bound to organic and inorganic ligands (Whiting and Clifford 1983). Urban stormwater pollution is by its nature sporadic and acts as a physical and chemical pulse on the receiving water ecosystem. Higher levels of urban pollutants, such as metals and hydrocarbons, are typically found during "flushing" storm events (Pitt et al. 1995). Also coincident with these elevated pollution level events is increased flow over the period of the storm. These "scouring," high-energy flows have been shown to have a negative synergistic impact on benthic populations (Borchardt and Statzner 1990). Some benthic species tend to migrate downstream or "drift" during stormflow conditions or pollutant events, while others try to avoid exposure by burrowing into the substrate.

One of the first comprehensive studies of the effects of urban runoff on benthic macroinvertebrates in streams was conducted on the East Coast (Garie and McIntosh 1986). This was a typical upstream (control) compared to downstream (impacted) site

study. Lead, zinc, and chromium were the predominant metals found in the stormwater. Macroinvertebrate diversity (number of taxa) and changes in community composition were used as the primary measures of impact. The results of this study again showed that there are both “tolerant” and “sensitive” species with regard to metal toxicity and urban runoff impact. The study also confirmed that elevated pollutant concentrations during urban runoff storm events were short-term and transient in nature, and it was hypothesized that the real impact on macroinvertebrate communities lay in long-term exposure to metals accumulating in the benthic sediments. This points out one of the potential flaws of using macroinvertebrates as biological surrogates for fish in that fish, unlike the benthos, are generally not exposed to toxic chemicals in sediments.

Another very comprehensive study conducted in the Pacific Northwest (PNW) showed that, although macroinvertebrate community structure was significantly changed due to urbanization impacts, the fish population structure of impacted and control streams remained largely the same (Pedersen and Perkins 1986). Apparently, salmonids feed on available benthos and do not select for specific trophic groups or species. This is not to say that a shift in benthic community structure is not a good indicator of urban impact, but one must be careful in extrapolating the results of one group of organisms to other biota, even if they are closely linked within the food web. The PNW study also demonstrated a lack of consistency when trying to use complex macroinvertebrate diversity indices to gauge the level of urban impact. Natural variability was generally too high and effectively masked any well-defined correlations.

Aquatic insect sampling and analysis has, however, been shown to be very useful as a tool for assessing other impacts of metal pollution (Clements 1994). The usefulness of benthic macroinvertebrates as monitors of bioavailable metal concentration and long-term bioaccumulation of metals has been demonstrated (Kiffney and Clements 1993). Still other studies have highlighted the synergistic negative impacts of metals and other habitat degradations on aquatic ecosystems in general (Hoiland and Rabe 1992). Finally, the persistence of sediment metal levels and resultant long recovery times has been shown for macroinvertebrate communities exposed to prolonged pollution inputs in the field (Chadwick et al. 1986).

At some point in their life cycle, many aquatic organisms have their principal habitat in, on, or near sediment. Examples of this include benthic macroinvertebrates that spend almost their entire larval stage in contact with sediments. In the PNW, salmonids also spend an extensive portion of their embryonic life stage within the benthic environment of their natal stream. In addition to functioning as benthic habitat, sediments can also capture and retain pollutants introduced by urban runoff. Pollutants enter sediments in several ways. The most direct path is the settling of suspended solids. Sediments deposited by urban runoff can physically degrade the substrata by filling interstitial spaces used as habitat by benthic organisms or by reducing DO transfer within the benthic environment. Dissolved pollutants can also move out of solution and into

sediments by such mechanisms as adsorption of metals and organics at the sediment surface, ion exchange of heavy metals in water with native calcium, magnesium, and other minerals in sediments, as well as the precipitation of phosphorus (Burton and Pitt 2002).

Most aquatic sediments have a large capacity to receive such contaminants through these processes. Also, many of the particulate pollutants are conservative. Once in the sediment, they do not decompose or significantly change form. These conservative pollutants include refractory organic chemicals relatively resistant to biodegradation as well as all metals. Consequently, these types of pollutants progressively accumulate in sediments. Over the long term, discharge of even modest quantities of pollutants can result in sediment concentrations several orders of magnitude higher than in the overlying water. These contaminant reservoirs can be toxic to aquatic life through direct contact. They can also spread beyond the benthos, and bio-magnify through the food web (Burton and Pitt 2002).

Historically, water quality has received more attention than sediment contamination. In the past ten to fifteen years, this approach has changed because of mounting evidence of environmental degradation in areas that meet water quality criteria. However, sediment toxicity investigations are limited because we lack accepted testing methods and do not fully understand the factors that control contaminant bioavailability. The result is an approach that emphasizes bioassay exposure techniques, either in situ or in the laboratory, along with chemical analysis of the sediments, overlying water, and/or sediment interstitial water. Very few studies have focused on the eco-toxicology of contaminated sediments in the natural environment (Chapman et al. 1998).

Water quality and nutrients

Watershed urbanization generally leads to higher nutrient (phosphorus and nitrogen) concentrations in stormwater runoff (Omernik 1976). Phosphorus is generally found in particulate form, but the more bioavailable, dissolved forms are also common. Nitrogen is typically found in the nitrate or ammonium form. Sources of nutrients in urbanizing catchments include lawn and garden fertilizers, wastewater (failing septic systems and sewage treatment plant discharges), and fine sediment from erosion or street runoff. Although nutrient pollution is often associated more with agricultural activities, urbanization can contribute significant quantities of nutrients to receiving waters (Omernik 1976).

Eutrophication is the process through which excess nutrients cause overall algal biomass increases, especially during “bloom” periods. This is due to increased loading of the nutrient that had previously been in shortest supply relative to need. In freshwater lakes, this limiting nutrient is most often phosphorus, and secondarily nitrogen. In estuarine or marine nearshore areas, nitrogen is typically the limiting nutrient. In

addition to promoting larger quantities of algae, nutrient enrichment typically changes the composition of the algal community. One-celled diatoms give way to filamentous green forms, followed by blue-green forms (some toxic) with a larger nutrient supply (Welch 1980; Welch et al. 1988; Welch et al. 1989; Welch et al. 1992).

As discussed earlier, urban areas have a number of nutrient sources, and nutrient loadings increase with the development level. Eutrophication degrades lake and estuarine ecosystems in several ways. The filamentous algae are poorer food than diatoms to herbivores because of their structure and, sometimes, bad taste and toxicity. Filamentous algae clog water intakes and boat propellers and form odorous masses when they wash up on beaches. They also reduce water clarity, further limiting beneficial uses. When a large biomass dies at the end of the bloom, its decomposition by bacteria creates high oxygen demand, which can result in severely depressed DO levels (Welch 1980; Shuster et al. 1986; Walker 1987). In addition to algal blooms and the associated negative impacts, eutrophication may result in an overall increase in other nuisance plants, including a variety of submerged or emergent aquatic macrophytes. Some of these plant communities may include invasive species such as hydrilla, Eurasian milfoil, purple loosestrife, and reed canary grass (Welch 1980).

Ecosystem processes: the habitat connection

Degradation of aquatic habitat is one of the most significant ecological impacts of the changes that accompany watershed urbanization. The complex physical effects from elevated urban streamflows, stream channel alterations, and riparian encroachment can damage or destroy stream and wetland habitats. In addition to the indirect effects of habitat degradation or loss, aquatic biota can be directly affected by the cumulative impacts of urbanization. Table 1 summarizes the effects of urbanization on aquatic ecosystems.

Biological degradation is generally manifested more rapidly than physical degradation. Aquatic biota tend to respond immediately to widely fluctuating water temperatures, water quality, reduced inputs of organic matter or other food sources, more frequent elevated streamflows, greater wetland water level fluctuations, or higher sediment loads. These stressors may prove to be fatal to some sensitive biota, impair the physiological functions of others, or encourage mobile organisms to migrate to a more habitable environment.

Ecological and biological effects of watershed urbanization include the following:

- Loss of instream complexity and habitat quality due to increase in bankfull flow frequency and duration.
- Reduced habitat due to channel modifications, and reduced baseflows, causing

crowding and increasing competition for refuge and foraging habitat.

- Shifts in populations and communities of environmentally sensitive organisms to biota more tolerant of degraded conditions. Reduced biota abundance and biodiversity.
- Scouring and washout of biota and structural habitat elements from urban stream channels.
- Sediment deposits on gravel substrates where fish spawn and rear young and where algal and invertebrate food sources live. Reduced survival of egg and embryonic life stages.
- Direct loss of habitat due to the replacement of natural stream channels and wetlands with engineered drainage channels and stormwater treatment ponds.
- Loss of ecologically functional pool-riffle habitat characteristics in stream channels. Loss of deep-water cover in rearing habitat and loss of spawning habitat.
- Aesthetic degradation and loss of recreational beneficial uses.
- Direct effects of suspended sediment on aquatic organisms, like abrasion of gills and other sensitive tissues, reduced light for photosynthesis, reduced visibility for catching food and avoiding predators, and transport of metallic, organic, oxygen-demanding, bacterial, and nutrient pollutants.
- Reduction in pool area and quality. Loss of refuge habitat for adult and juvenile fish.
- Loss of riparian vegetation, resulting in stream bank erosion, loss of shading and temperature regulation, reduced leaf-litter and organic matter input, loss of overhanging vegetation cover, and reduced LWD recruitment.
- Loss of LWD function, including hydraulic roughness, habitat formation, and refugia habitat.
- Increased summer temperatures because of lower baseflow and less water availability for heat absorption. Decline in DO from the lower oxygen solubility of warmer water.
- Less dilution of pollutants as a result of lower baseflows, which in turn results in higher concentrations and shallower flow that can interfere with fish migrations and localized movements.
- Increased inorganic and organic pollutant loads with potential toxicity impacts.
- Increased bacterial and pathogen pollution, which can result in an increase in disease in aquatic biota and humans.
- Elevated nutrient loading and resultant eutrophication of lake, wetland, and estuarine habitats. Reduced DO as a possible result of eutrophic conditions, which in turn reduces usable aquatic habitat.
- More barriers to fish migration, such as blocking culverts and diversion dams.
- Overall loss of habitat quality, complexity, and diversity due to channel and floodplain simplification or loss.

Table 1 - Summary of the impacts of urbanization on aquatic ecosystems.

Environmental Concern	Potential Impact	Cause - Source
Increase in Runoff-Driven Peak or Bankfull Stream Flows	Degradation of aquatic habitat and/or loss of sensitive species	Increased stormwater runoff volume due to an increase in basin imperviousness
Increase in Runoff-Driven Flooding Frequency & Duration	Degradation of aquatic habitat and/or loss of sensitive species	Increased stormwater runoff volume due to an increase in basin imperviousness
Increase in Wetland Water Level Fluctuations	Degradation of aquatic habitat and/or loss of sensitive species	Increased stormwater runoff due to an increase in basin imperviousness
Decrease in Dry Season Baseflows	Reduced aquatic habitat and less water for human consumption, irrigation, or recreational use	Water withdrawals and/or less natural infiltration due to an increase in basin imperviousness
Streambank Erosion and Stream Channel Enlargement	Degradation of aquatic habitat and increased fine sediment production	Increase in stormwater runoff driven stream flow due to an increase in basin imperviousness
Stream Channel Modification due to Hydrologic Changes and Human Alteration	Degradation of aquatic habitat and increased fine sediment production	Increase in stormwater runoff driven stream flow and/or channel alterations such as levees and dikes
Streambed Scour and Incision	Degradation of aquatic habitat and loss of benthic organisms due to washout	Increase in stormwater runoff driven stream flow due to an increase in basin imperviousness
Excessive Turbidity	Degradation of aquatic habitat and/or loss of sensitive species due to physiological and /or behavioral interference	Increase in stormwater runoff driven stream flow and subsequent streambank erosion due to an increase in basin imperviousness
Fine Sediment Deposition	Degradation of aquatic habitat and loss of benthic organisms due to fine sediment smothering	Increase in stormwater runoff driven stream flow and subsequent streambank erosion due to an increase in basin imperviousness
Sediment Contamination	Degradation of aquatic habitat and/or loss of sensitive benthic species	Stormwater runoff pollutants
Loss of Riparian Integrity	Degradation of riparian habitat quality and quantity, as well as riparian corridor fragmentation	Human development encroachment and stream road crossings
Proliferation of Exotic & Invasive Species	Displacement of natural species and degradation of aquatic habitat	Encroachment of urban development

Elevated Water Temperature	Lethal and non-lethal stress to aquatic organisms – reduced DO levels	Loss of riparian forest shade and direct runoff of high temperature stormwater from impervious surfaces
Low Dissolved Oxygen (DO) Levels	Lethal and non-lethal stress to aquatic organisms	Stormwater runoff containing fertilizers and wastewater treatment system effluent
Lake & Estuary Nutrient Eutrophication	Degradation of aquatic habitat and low DO levels	Stormwater runoff containing fertilizers and wastewater treatment system effluent
Bacterial Pollution	Human health (contact recreation and drinking water) concerns, increases in diseases to aquatic organisms, and degradation of shellfish harvest beds	Stormwater runoff containing livestock manure, pet waste, and wastewater treatment system effluent
Toxic Chemical Water Pollution	Human health (contact recreation and drinking water) concerns, as well as bioaccumulation and toxicity to aquatic organisms	Stormwater runoff containing toxic metals, pesticides, herbicides, and industrial chemical contaminants
Reduced Organic Matter (OM) & Large Woody Debris (LWD)	Degradation of aquatic habitat and loss of sensitive species	Loss or degradation of riparian forest and floodplain due to development encroachment
Decline in Aquatic Plant Diversity	Alteration of natural food web structure and function	Cumulative impacts of urbanization
Decline in Aquatic Invertebrate Diversity	Alteration of natural food web structure and function	Cumulative impacts of urbanization
Decline in Amphibian Diversity	Loss of ecologically important species	Cumulative impacts of urbanization
Decline in Fish Diversity and Abundance	Loss of ecologically important species	Cumulative impacts of urbanization

Numerous studies have documented the effect of watershed urbanization on the degradation of instream habitat and the decline of native biota. These include research from almost all parts of the United States and from developed countries around the world. The earliest research efforts to study the cumulative impacts of urbanization on small-stream habitat and stream biota were conducted in the Puget Sound region (Richey, 1982; Scott, 1982; Steward, 1983) and in the Chesapeake Bay region (Ragan and Dietermann, 1975; Ragan et al., 1977; Klein, 1979). These were followed by even more comprehensive studies in the same regions and in other parts of the country. This section describes the findings of this body of research.

Table 2 - Summary of research on urban stream habitat, water-quality, and biota.

Research Study	Habitat	WQ	Fish	Macroinvertebrates	Location
Ragan & Dietermann, 1975		X	X		MD
Klein, 1979	X	X	X		MD
Richey, 1982	X				WA
Pitt and Bozeman, 1982		X	X	X	CA
Steward, 1983			X		WA
Scott et al., 1986	X		X		WA
Jones and Clark, 1987		X		X	VA
Steedman, 1988				X	OT
Limburg & Schmidt, 1990		X	X		NY
Schueler & Galli, 1992			X		DC
Booth & Reinelt, 1993	X				WA
Lucchetti & Fuerstenberg, 1993			X		WA
Black & Veatch, 1994	X		X	X	MD
Weaver & Garman, 1994			X		VA
Lenat & Crawford, 1994	X	X	X	X	NC
Galli, 1994	X		X		DC
Jones et al., 1996	X	X		X	VA
Hicks & Larson, 1997	X				MA
Booth & Jackson, 1997	X				WA
Kemp & Spotila, 1997			X	X	PA
Maxted & Shaver, 1997	X			X	DE
May et al., 1997	X	X	X	X	WA
Wang et al., 1997	X		X		WI
Harding et al., 1998	X		X	X	NC
Horner & May, 1999	X		X	X	WA
Kennen, 1999		X		X	NJ
MNCPPC, 2000	X		X	X	MD
Finkenbine et al., 2000	X				BC
Meyer & Couch, 2000	X		X	X	GA
Wang et al., 2000	X		X		WI
Horner et al., 2001	X		X	X	WA/TX/MD
Nerbonne & Vondracek, 2001	X		X	X	MN
Stranko & Rodney, 2001	X				MD
Wang et al., 2001	X		X		WI
Morse et al., 2003	X	X		X	ME

One of the most common effects of watershed urbanization on instream habitat is the loss of habitat quality, diversity, and complexity. This is the so-called “simplification” of urban stream characteristics. In undisturbed, properly functioning stream systems, the

natural (mainly hydraulically driven) disturbance regime maintains the stream in a state of dynamic equilibrium. This means that the stream ecosystem is stable, but not static. Changes occur on several spatial and temporal time scales.

These changes can be small and subtle, such as a riparian tree falling into a creek (LWD recruitment) and forming a new pool habitat unit as the result of the hydro-geomorphic interaction of the streamflow and the LWD. Changes can also be large and catastrophic, such as those occurring during major flooding events that can rearrange the entire channel form of a stream system. Natural streams tend to have a level of redundancy and complexity that allows them to be resilient in responding to disturbance. Streams may change over time as a result of natural habitat-forming processes (flooding, fire, LWD recruitment, sediment transport, OM and nutrient cycling, and others), but they continue to support a complex stream-riparian ecosystem and a diverse array of native biota.

As mentioned above, the first Puget Sound stream research project compared ecological and biological conditions in an urbanized stream (Kelsey Creek) and a relatively natural stream (Big Bear Creek). Urbanized Kelsey Creek was found to be highly constrained by the encroachment of urban development, with 35 percent of the stream banks armored with rip-rap, and the floodplain-riparian zone also highly modified. Bear Creek, on the other hand, had less than 10 percent stream bank armoring and a natural riparian corridor and channel migration zone. Road-crossing bridges and culverts were frequent on Kelsey Creek, but not on Bear Creek (Richey 1982). Large woody debris and other natural habitat complexity features common in Bear Creek were also lacking in Kelsey Creek (Steward 1983).

In the Puget Sound comparison of urban and non-urban streams, Kelsey Creek, an urban stream, experienced twice the bed scour of its non-urban counterpart (Scott 1982). As a consequence, sediment transport was three times as great in Kelsey Creek (Richey 1982) and fines were twice as prevalent in its substrates (Scott 1982). The invertebrate communities in different benthic locations produced 14 to 24 taxa in Bear Creek but only six to 14 in Kelsey Creek (Richey 1982). Salmonid fish diversity also differed. Bear Creek had four salmonid species of different age-classes, whereas Kelsey Creek had only one non-anadromous species mainly represented by the 0- to 1-year age class (Scott 1982 and Steward 1983). Although we cannot explicitly determine the relative roles of hydrology and habitat quality, much evidence shows that hydrologic alteration and the related sediment transport were most responsible for the biological effects (Richey 1982).

Several studies in the Pacific Northwest examined various aspects of the influence of urban hydrology on salmon and salmon habitat. Data show a significant decrease in young salmon survival in both large and small streams when flow events occur that are equal to or larger than the natural five-year frequency discharge. Since the frequency of such events increases tremendously after urbanization, salmonids experience great

difficulty in urban streams. These investigations also pointed out the relationship between urbanization level and biological integrity. The study rated channel stability along numerous stream reaches and related it to the proportion of impervious areas within the watershed. Stability was significantly higher where imperviousness was less than 10 percent (Booth and Reinelt 1993). The study rated habitat quality along streams in two basins according to four standard measures. Marked habitat degradation occurred at 8 to 10 percent total impervious area (TIA). Population data on cutthroat trout and less tolerant coho salmon from streams draining nine catchments did not show a distinct threshold. They indicated, however, that population shifts are measurable with just a few percent of impervious area and become substantial beyond about 10 to 15 percent (Lucchetti and Fuerstenberg 1993). Later studies in the same region confirmed this decline in salmonid abundance and diversity, as well as the degradation of salmon habitat at very low levels (5 to 10 percent TIA) of imperviousness in small urban streams (May 1997; May et al. 1997; Horner and May 1999).

More recent research projects in the Puget Sound region (May et al. 1997) and in Vancouver, British Columbia (Finkenbine et al. 2000) found that the degradation of instream and riparian habitat quality, diversity, and complexity are common features of urban streams. There appears to be a linear decline in most measures of habitat quality in relationship to the level of watershed urbanization or imperviousness. Instream LWD, which is a critical habitat complexity element in streams in forested watersheds, tends to become scarce when TIA approaches the 10 to 20 percent range (May et al. 1997; Horner et al. 1997; Finkenbine et al. 2000). Streambed quality also declines as urbanization increases (May et al., 1997; Horner et al., 1997; Finkenbine et al. 2000). This decline in benthic habitat is typically characterized by higher levels of fine-sediment deposition, substrata embeddedness, streambed coarsening, and frequent streambed scour events.

Similar to these studies in the Pacific Northwest, Morse (2003) observed that both instream habitat and water quality in small urbanizing streams in Maine declined in a linear fashion. Studies in Delaware (Maxted and Shaver 1997), Wisconsin (Wang et al. 1997), and Minnesota (Nerbonne and Vondracek 2001) confirm this trend. These findings have also been replicated in other countries, most notably in Australia (Davies et al. 2000) and New Zealand (Allibone et al. 2001).

This simplification of the stream channel and loss of instream habitat complexity results in a restructuring of the stream fish community in the urbanized creek. Urban impacts had a much greater impact on coho salmon (*Oncorhynchus kisutch*) than on cutthroat trout (*Oncorhynchus clarki*), which appear to be more tolerant of urban stream conditions (Scott et al., 1986). Pitt and Bissonnette (1984) and Lucchetti and Fuerstenberg (1993) also found similar results in other studies of streams in the Puget Sound lowland eco-region. Coho salmon, which normally out-compete cutthroat trout in natural streams, appear to be more sensitive to changes associated with urbanization and therefore decline in abundance as urban development increases (May 1997; May et al. 1997; Horner et al.

1997; Horner and May 1999). Figure 7 illustrates the shift in salmonid species found in urbanizing streams in the Puget Sound lowland eco-region.

Ragan and Dietermann (1975) attributed the loss of fish species diversity in urban streams in the Chesapeake eco-region of Maryland to the cumulative effects of urban development. A study in Ontario, Canada (Steedman 1988) also found a shift in fish community structure due to the cumulative impacts of watershed land use and riparian corridor encroachment. Similar results were seen for fish community structures in New York (Limburg and Schmidt 1990), Virginia (Weaver and Garman 1994), Pennsylvania (Kemp and Spotila 1997), North Carolina (Harding et al. 1998), and Georgia (Gillies et al. 2003).

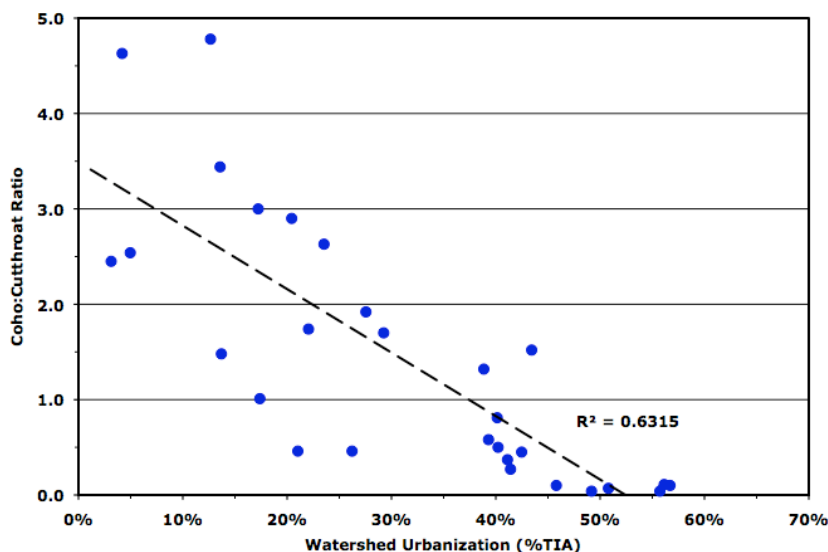


Figure 7 – Effect of Urbanization on Fish in the Puget Sound Region (from May et al. 1997)

A study in Mississippi found that instream habitat quality in urbanizing stream channels impacted by high-flow incision was significantly inferior to the quality of reference stream channels in undeveloped watersheds. In addition, the reference streams had greater mean water depths, more channel complexity in the form of woody debris, and more deep pool refuge habitat than the impacted streams. Relative to the reference streams, fish assemblages in the incised stream channels were composed of smaller fish and fewer species (Shields et al. 1994).

In several extensive studies of urbanizing streams in Wisconsin, a significant relationship was found between watershed land use and instream habitat as well as stream fish communities (Wang et al. 1997; Wang et al. 2000; Wang et al. 2001). In these studies, stream fish abundance and diversity both declined as watershed development increased above the 8 to 12 percent total impervious range. These studies also compared agricultural impacts to urban impacts, finding that urbanization had more severe and longer lasting effects. Habitat destruction and water-quality degradation were found to be the main contributing factors to the overall decline in stream ecosystem health. In addition, natural riparian vegetation (buffer) conditions had a significant influence on instream habitat conditions and appeared to at least partially mitigate some of the negative impacts of watershed urbanization (Wang et al. 2001).

A study in Washington, DC (Galli 1991) investigated the local thermal impacts of urban runoff on stream ecosystems and reached the following conclusions:

- Air temperature was the strongest influence on stream water temperature.
- Average stream temperature increased linearly with stream sub-basin imperviousness.

- Some temperature criteria violations occurred just above 10 percent TIA and increased in severity and frequency with more imperviousness.
- All tested structural stormwater treatment facilities under best management practice (BMP) that had a surface discharge caused some violations of temperature criteria under both baseflow and storm runoff conditions.
- Based on the findings from a literature review, the investigators concluded that the thermal conditions produced by urban runoff and treatment facilities could cause succession from cold-water diatoms to warm-water filamentous green and blue-green algal species, as well as severe impacts on cold-water invertebrates and fish. A shift from cold-water community composition to warm-water organisms and exotic species is very possible in highly urbanized watersheds.

It should be noted that the life cycles of native fish can differ significantly even among closely related species. Attention must be paid to the life history specifics and habitat requirements of the various species of concern in the urban watershed being managed before any decisions are made on conservation, restoration, or mitigation of stormwater runoff impacts. Different fish carry out their migrations, reproduction, and rearing at different times and have freshwater stages of various lengths. Management must ensure that all life stages (egg, embryonic, juvenile, and/or adult) have the habitat conditions needed at the right time and that no barriers to migration exist.

Ohio has an extensive database relating watershed development and land use to fish abundance and diversity. These data suggest that there are multiple levels of fish response to increasing urbanization. At the rural level of development (under 5 percent urban land use), sensitive species begin to disappear from streams. In the 5 to 15 percent urban land-use range (suburban development), habitat degradation is common and fish continue to decline in abundance and diversity. In addition, aquatic invertebrates also decline significantly. Above 15 percent watershed urbanization, habitat degradation, toxicity effects from physico-chemical water pollution, and nutrient enrichment result in severe degradation of fish fauna (Yoder et al., 1999). There have been similar findings in studies in Alabama (Onorato et al. 2000) and North Carolina (Lenat and Crawford 1994).

The cumulative effects of urbanization, including altered hydrologic and sediment transport regimes as well as channel modifications and degraded instream habitat, were also found to cause a shift in the aquatic insect communities of urban streams in the Puget Sound region (Pedersen and Perkins 1986; May et al. 1997; Horner and May 1999; Morley and Karr 2002). This relationship between watershed urbanization, stormwater runoff pollution, and aquatic insect community taxonomic composition has also been observed in small stream studies in northern Virginia (Jones and Clark 1987; Jones et al. 1996), Pennsylvania (Kemp and Spotila 1997), New Jersey (Kennen 1999), and Maine (Morse 2003). These findings have also been replicated in other countries, most notably in Australia (Walsh et al. 2001) and New Zealand (Collier and Winterbourn 2000).

Aquatic insects and other macroinvertebrates have been found to be useful indicators of environmental conditions in that they respond to changes in natural land cover and human land use (Black et al. 2004). Overall, there tends to be a decline in taxa richness or species diversity, a loss of sensitive species, and an increase in tolerant species (such as chironomids) due mainly to the cumulative impacts of watershed urbanization: altered hydrologic and sediment transport regimes, degradation of instream habitat quality and complexity, stream bed fine sediment deposition, poor water quality, and the loss of native riparian vegetation. In many cases, the many aquatic insects and benthic macroinvertebrates sampled from streams or wetlands are combined into a set of indices to standardize comparisons between stream samples. Often the mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera) are combined into an “EPT” index. In some cases, multi-metric indexes have been developed that include several measures of the characteristics of the stream macroinvertebrate community. The EPA Rapid Bioassessment Protocol (RBP) and the Benthic Index of Biotic Integrity (BIBI) are examples of this (Karr 1998). Figure 8 illustrates the BIBI scores for urbanizing streams in the Puget Sound lowland eco-region.

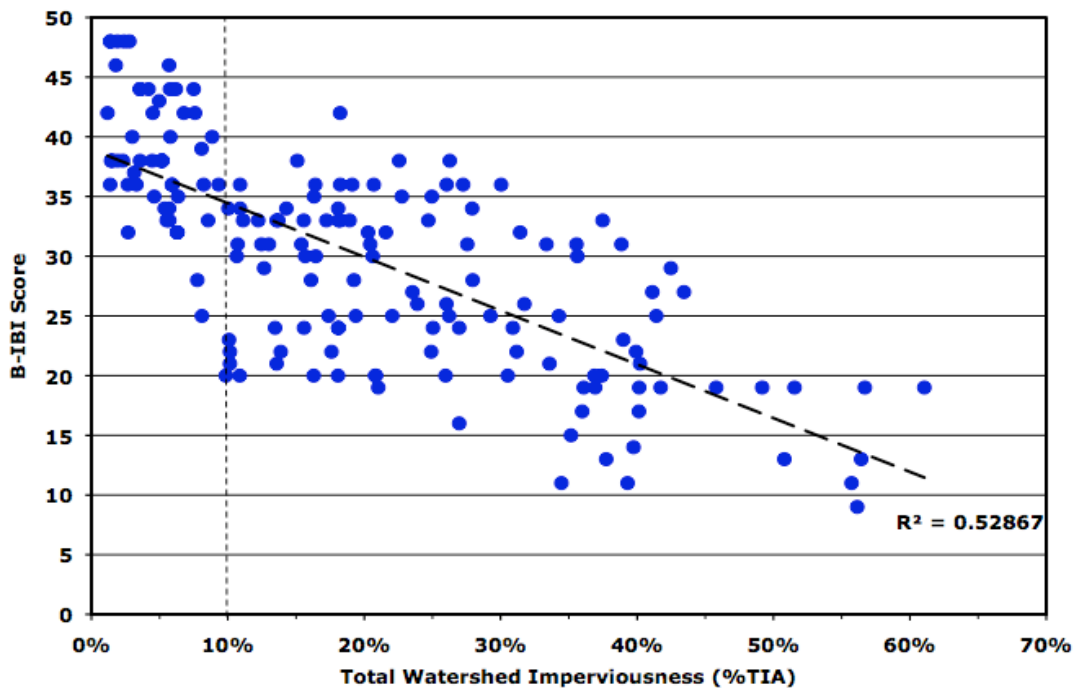


Figure 8 – Effect of Urbanization on Stream Biota in the Puget Sound Region (from May et al. 1997)

Ecosystem processes and riparian habitat

Riparian vegetation or the streamside forest is an integral component of all stream ecosystems. This is especially true of forested regions of the Pacific Northwest (PNW). A wide, nearly continuous corridor of mature forest, off-channel wetlands, and complex floodplain areas characterizes the natural stream-riparian ecosystems of the PNW (Naiman and Bilby 1998). Native riparian forests of the region are typically dominated by a complex, multi-layered canopy of mature conifers, mixed with patches of alder, where disturbance has occurred in the recent past (Gregory et al. 1991). The riparian forest also includes a complex, dense, and diverse understory and ground cover vegetation. In addition, the extensive upper soil layer of forest “duff” provides vital water retention and filtering capacity for the ecosystem. A typical natural riparian corridor in the Puget Sound lowlands also includes a floodplain area, a channel migration zone (CMZ), and numerous off-channel wetlands. Natural floodplains, an unconstrained CMZ, and complex riparian wetlands are critical components of a properly functioning aquatic ecosystem (Naiman and Bilby 1998). Organic debris and vegetation from riparian forests also provide a majority of the organic carbon and nutrients that support the aquatic ecosystem food web in these small lowland streams. In short, the riparian community (vegetation and wildlife) directly influences the physical, chemical, and biological conditions of the aquatic ecosystem. Reciprocally, the aquatic ecosystem affects the structure and function of the riparian community.

In addition to the characteristics of the riparian forest described above, the most commonly recognized functions of the riparian corridor include the following:

- Providing canopy-cover shade necessary to maintain cool stream temperatures required by salmonids and other aquatic biota. Regulation of sunlight and microclimate for the stream-riparian ecosystem (Gregory et al. 1991).
- Providing organic debris, leaf litter, and other allochthonous inputs that are a critical component of many stream food webs, especially in headwater reaches (Gregory et al. 1991; Naiman et al. 2000; Rot et al. 2000).
- Stabilizing stream banks, minimizing stream bank erosion, and reducing the occurrence of landslides while still providing stream gravel recruitment (Naiman et al. 2000).
- Interacting with the stream channel in the floodplain and channel migration zone (CMZ). Retention of flood waters. Reduction of fine sediment input into the stream system through floodplain sediment retention and vegetative filtering (Naiman et al. 2000).
- Facilitating the exchange of groundwater and surface water in the riparian floodplain and stream hyporheic zone (Correll et al. 2000).
- Filtering and vegetative uptake of nutrients and pollutants from groundwater and stormwater runoff (Fischer et al. 2000).

- Providing recruitment of large woody debris (LWD) into the stream channel. LWD is the primary in-stream structural element and functions as a hydraulic roughness element to moderate streamflows. LWD also serves a pool-forming function, providing critical salmonid rearing, flow refugia, and enhanced instream habitat diversity (Fetherston et al. 1995; Rot 1995; Rot et al. 2000).
- Providing critical wildlife habitat including migration corridors, feeding and watering habitat, and refuge habitat (Gregory et al. 1991; Fischer et al. 2000; Hennings and Edge 2003). Providing primary habitat for aquatic habitat modifiers such as beaver and many other terrestrial predators or scavengers associated with salmonids.

Based on the results of research in the Puget Sound region (May et al. 1997), the term *riparian integrity* was adopted to describe the conditions found in natural lowland stream-riparian ecosystems. These properly functioning conditions can serve as a template for evaluation and management of riparian areas. As used here, riparian integrity includes both structural and functional elements characteristic of the natural stream-riparian ecosystem. Land-use activities and development encroachment pressure can have a negative impact on native riparian forests and wetlands, which are intimately involved in stream ecosystem functioning. Riparian integrity includes the following components:

- Lateral riparian extent (so-called “buffer” width);
- Longitudinal riparian corridor connectivity (low fragmentation);
- Riparian quality (vegetation type, diversity, and maturity); and
- Floodplain and channel migration zone (CMZ) integrity.

In general, urban riparian buffers have not been consistently protected or well managed (Schueler 1995; Wenger 1999; Horner and May 1999; Moglen 2000; Lee et al. 2004). This is certainly true of the Puget Sound region (Figure 9). Several factors reduce the effectiveness of riparian buffers in urbanizing watersheds. The surrounding land use may overwhelm the buffer, and human encroachment continues to occur in spite of established buffer zones. Buffers that are established by regulation during the construction phase of development are rarely monitored by jurisdictional agencies. Over the long term, oversight and management of buffer areas is often taken on by property owners, who frequently are not familiar with the purpose or proper maintenance of the buffer (Booth 1991; Schueler 1995; Booth et al. 2002).

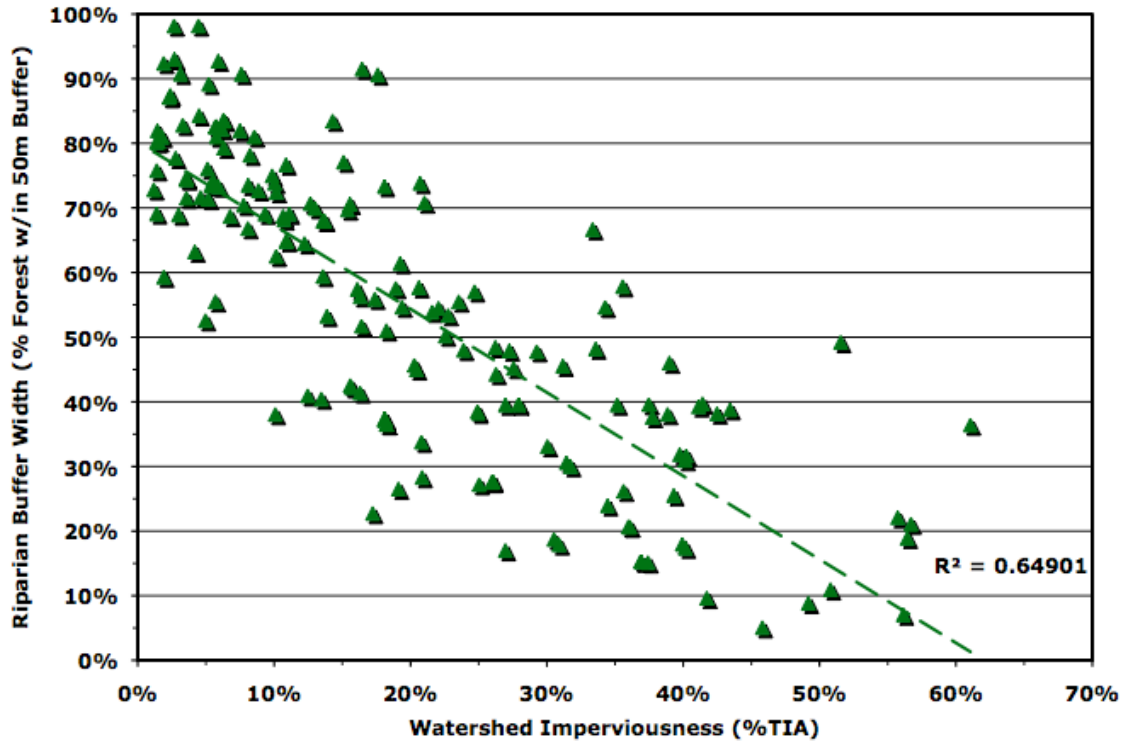


Figure 9. Impact of urbanization encroachment on riparian integrity (from May et al. 1997)

Ideally, the riparian corridor in a developing or developed watershed should mirror that found in the natural ecosystems of that region. Due to the cumulative impacts of past and present land use, this is often not the case (Figure 10). One example of this is the fragmentation of riparian corridors by roads, utility crossings, and other man-made breaks in the corridor continuity (Figure 11). Results from studies in the PNW and other regions indicate that streams with a high level of riparian integrity have a greater potential for maintaining natural ecological conditions than streams with urbanized riparian corridors (May and Horner 2000; Hession et al. 2000; Snyder et al. 2003). However, buffers can provide only partial mitigation for urban impacts on the stream-riparian ecosystem. At some point in the development process, upland urbanization and the accompanying disturbance is likely to overwhelm the ability of buffers to mitigate for urban impacts.

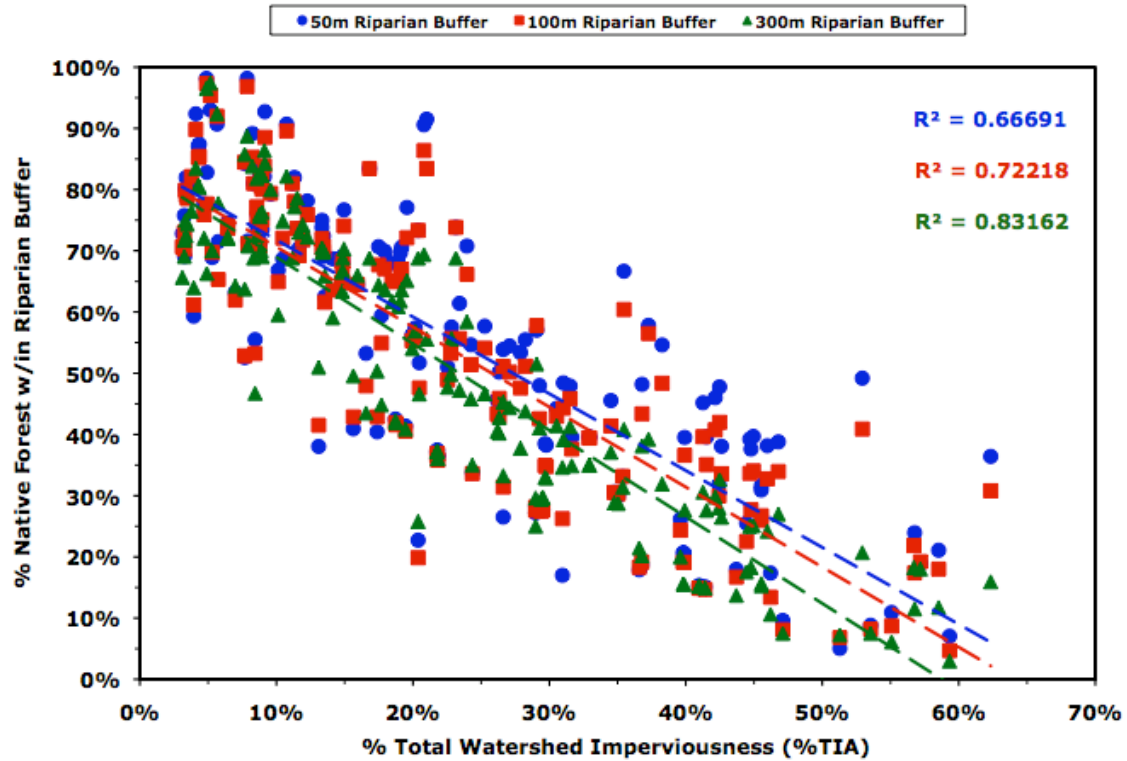


Figure 10. Impact of urbanization encroachment on riparian integrity (from May et al. 1997)

There are certain problems associated with the loss of functional riparian floodplain corridors around streams in urbanizing watersheds. These include changes in food web dynamics, higher stream temperatures, loss of instream habitat complexity (LWD), invasive species, stream bank erosion and greater inputs of sediment, excessive nutrient inputs, inflows of anthropogenic pollutants, and loss of wildlife habitat.

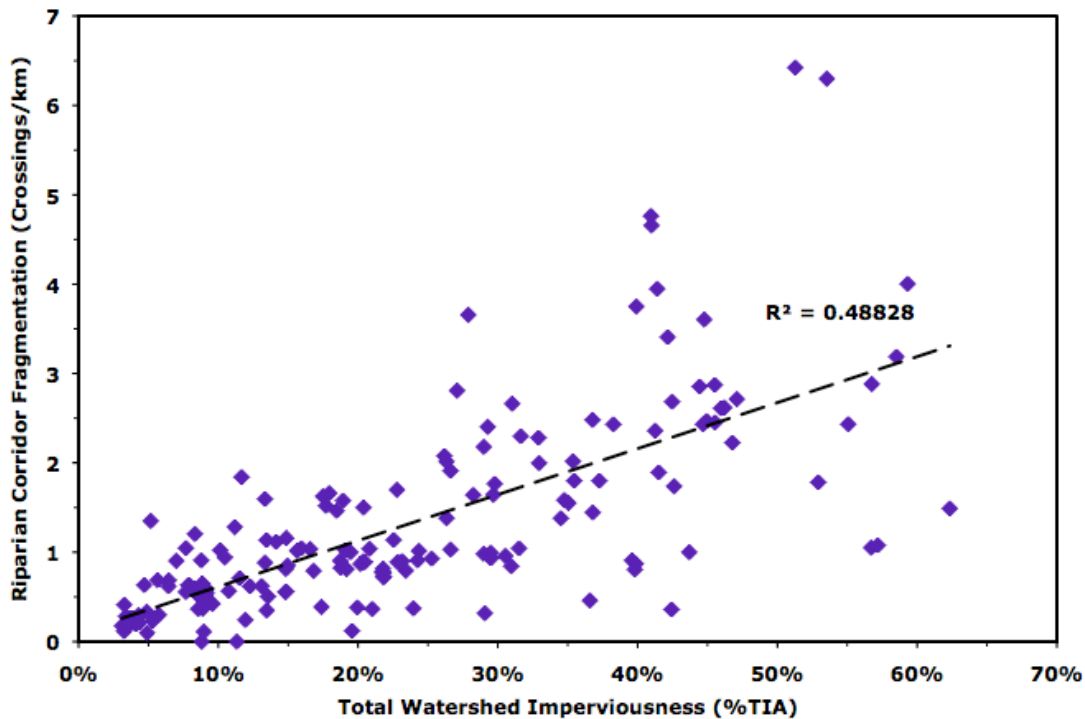
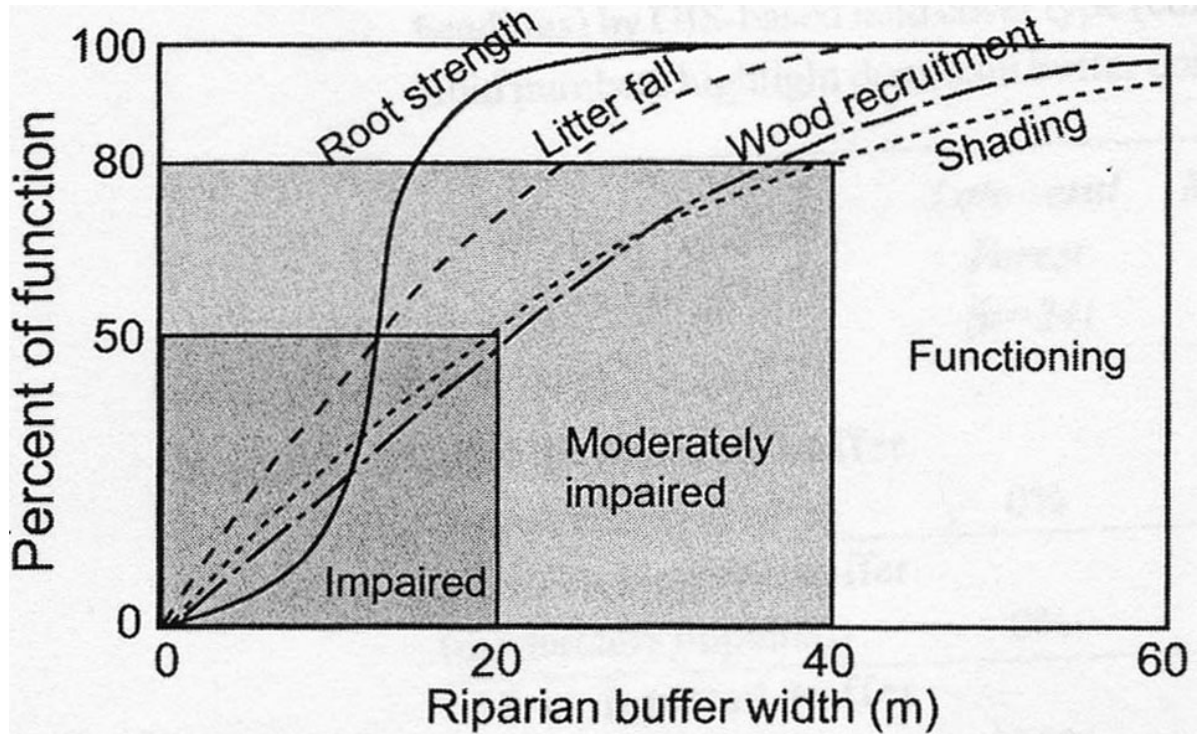


Figure 11 – Riparian corridor fragmentation and watershed urbanization (from May et al. 1997)

Stream temperature is regulated mainly by the amount of shade provided by the riparian corridor. This is an important variable affecting many instream processes such as the saturation value for dissolved oxygen (DO) in the water, organic matter decomposition, fish egg and embryonic development, and invertebrate life history (Paul and Meyer 2001). Removal of riparian vegetation, reduced groundwater recharge, and the “heat island” effect associated with urbanization all can increase water temperature in streams, lakes, rivers, wetlands, and nearshore marine areas. Invasive or exotic plants are another problem common to urban stream and wetland buffers. Human encroachment and landscaping activities can introduce exotic or invasive species into the riparian zone. These plants often out-compete native species, which can result in nuisance levels of growth.

Based on our current level of knowledge, the extent and configuration of urban riparian corridor buffers needed to protect the natural structure and function of the stream-riparian ecosystem cannot be described using a simple formula. Because of regional, watershed-scale, and site-level differences, as well as political issues, this is a fairly complex problem. The ecological and socio-economic value of the resource being protected should be considered when a riparian buffer or management zone is established. In addition, the local watershed, site, and riparian vegetation characteristics must be considered as well. The type and intensity of the surrounding land use should

also be factored into the equation so that some measure of physical encroachment and water-quality risk is made. Finally, the riparian functions that need to be provided should be evaluated. Figure 12 illustrates how this might be done (Femat 1993).



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