

Science Question 1: How does urbanization alter aquatic ecosystems and what are the implications for salmonid rehabilitation?

Urban and rural-residential development involves a diverse collection of actions that transforms natural ecosystems into highly altered landscapes. The suite of aquatic ecosystem changes that accompanies development have been reviewed by several authors (e.g., Paul & Meyer 2001; Brown *et al.* 2005b; Walsh *et al.* 2005b; Chin 2006; Kaye *et al.* 2006). In this review, the IMST summarizes changes most relevant to Pacific Northwest aquatic ecosystems with an emphasis on how these alterations affect native aquatic biota, particularly salmonids.

Streams, rivers and estuaries capable of supporting healthy salmonid populations contain a range of environmental attributes that meet the physiological and behavioral requirements of salmonids throughout their complex life cycles (Table 1-5). These environmental attributes typically reflect the historical conditions under which salmonid populations evolved unique sets of adaptations (i.e., life histories) that maximize their survival (Miller & Brannon 1982; Bjornn & Reiser 1991; Groot & Margolis 1991; Thorpe 1994; Quinn & Myers 2004). The spatial scales over which necessary environmental attributes are distributed are species specific. For example, chum salmon complete the freshwater stages of their life cycle over much smaller spatial scales than Chinook salmon (Figures 1-2 through 1-8 and Table 1-5 in Section 1.0). As a result, habitat alterations stemming from development may affect individual salmonid species differently.

IMST has identified four key pathways (hydrology, physical habitat, water quality, fish passage) through which urban and rural-residential developments alter aquatic ecosystems (Figure 2-1). IMST's intent in this review is to provide a general perspective of the importance of these factors to the goals of the Oregon Plan. While each pathway is addressed separately in this report, it is important to keep in mind that they occur simultaneously and exhibit strong interdependency at multiple spatial and temporal scales (King *et al.* 2005; also see reviews by Paul & Meyer 2001; Walsh *et al.* 2005b). In particular, the combined changes in hydrology, water quality, fish passage, and physical habitat ultimately lead to changes in aquatic biota, including salmonids (Figure 2-1). Because these pathways influence one another, and because the abiotic factors linking these pathways to aquatic ecosystem change are numerous, the specific mechanisms that underlie aquatic ecosystem impairments are not easily isolated and may vary with location or time of year. Effective rehabilitation of salmonid habitat in urban and rural-residential areas requires watershed-scale consideration of all pathways and their interactions (e.g., Booth *et al.* 2004; Walsh *et al.* 2005b).

Stormwater and wastewater are two major products of urban and rural-residential development that contribute to alterations in hydrologic processes, physical habitat, water quality, and fish passage. To highlight the importance of these two factors and to reduce repetition throughout this report, the following sections summarize the challenges stormwater and wastewater present to aquatic ecosystem functions in Oregon's streams, rivers and estuaries.

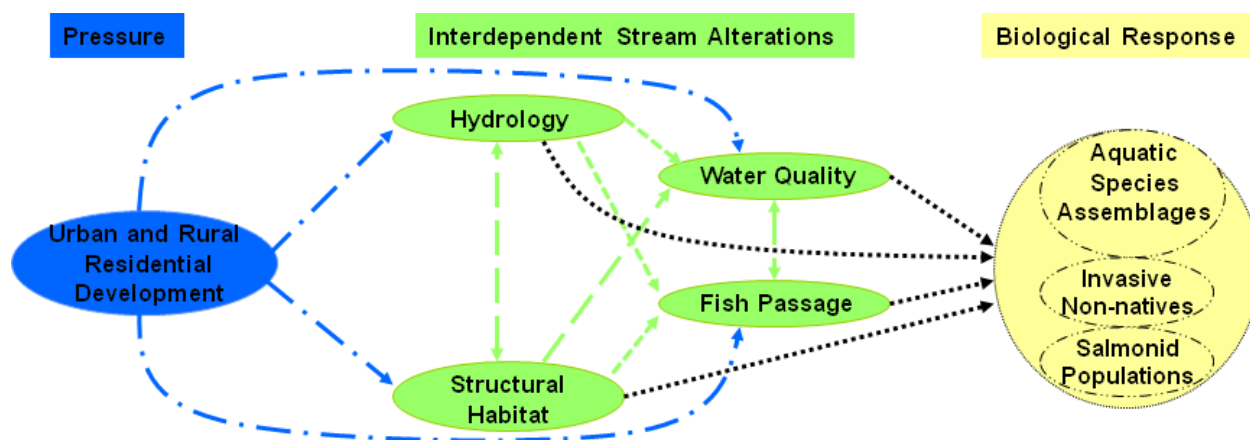


Figure 2-1. The pathways by which urban and rural-residential development alter aquatic communities and the condition of salmonid habitat and populations. Dashed/dotted (blue) arrows indicate direct effects of development on four key components of aquatic ecosystems (green). Dashed (green) arrows depict interactions among components, and dotted (black) arrows represent the influence of the altered ecosystem components on aquatic biota.

Section 2.0: Stormwater

Urban and rural-residential development causes profound changes to the pathways, volume, timing, and composition of stormwater runoff (Paul & Meyer 2001; Konrad & Booth 2005; Walsh *et al.* 2005b). These changes have diverse and persistent consequences for aquatic ecosystems that can extend throughout entire watersheds (Walsh *et al.* 2005a, 2005b). The State of Washington identified stormwater as a major cause of impairment in salmonid streams flowing through developed areas (GSRO 1999; ISP 2003). The extent of aquatic ecosystem impacts is strongly influenced by development pattern and stormwater infrastructure design (Dietz 2007; Walsh 2000; Walsh *et al.* 2005a). The suite of watershed-scale perturbations associated with changes in stormwater runoff includes physical, chemical, and biological changes.

Physical Changes: Development activities, that remove vegetation, compact soil, and/or create pavement and roofs, increase watershed imperviousness to precipitation. Consequently, the fraction of precipitation routed to surface runoff increases, triggering higher peak streamflows, the possibility of lower base flows, and greater flow variability (see Section 4.0 of this report). Disturbed vegetation and soil, and increased surface runoff, work in combination to alter watershed sediment budgets, which can produce a number of stressors for many aquatic organisms. These effects alter stream hydraulics, channel complexity, and channel morphology in ways that impair physical habitat (see Section 5.0 of this report) required by native aquatic biota.

Chemical Changes: Pollutants carried by stormwater runoff are a significant threat to freshwater resources (USEPA 1995). Water running over paved surfaces and through storm drains often bypasses soil and riparian vegetation that can buffer acidity, filter, and break down pollutants before they enter streams (Gresens *et al.* 2007). Stormwater frequently contains automotive

products (i.e., gasoline, oil, and antifreeze), metals, pesticides, road and airport deicers, fertilizers, and contaminated sediments (USEPA 2002b; Paul & Meyer 2001; Nielson & Smith 2005). Many heavy metals and pesticides are highly toxic to aquatic organisms and can alter the biotic communities of rivers and estuaries (Graves *et al.* 2004). Even at relatively lower rural-residential land use densities, untreated substances enter surface waters with negative consequences for aquatic biota (see Section 7.6 of this report).

Biological Changes: The changes in stormwater runoff associated with development can alter hydrologic processes, water quality, and physical habitat in ways that are not only detrimental to aquatic life but are also disproportionate to their land area (see reviews by Paul & Meyer 2001; Allan 2004; Booth 2005; Brown *et al.* 2005b; Walsh *et al.* 2005b). Numerous studies relating macroinvertebrate indices or fish assemblage composition to measures of urbanization typically show deteriorating conditions as urbanization increases (see Section 8.0 of this report). At the landscape scale, fish assemblages become increasingly homogeneous¹⁴ as development increases. This pattern is in part due to the physical habitat impairments associated with stormwater runoff. Increased stormwater runoff can limit salmonid population productivity or persistence by reducing survival, altering behavior, slowing growth, or delaying transitions between different developmental stages (see Section 8.3 of this report).

Section 2.1: Stormwater Management

Early efforts to manage stormwater typically used drainage infrastructure to move excess runoff away from people, structures, and transportation systems directly into streams that served as stormwater channels (Walsh 2000; Carter & Rasmussen 2006; Zheng *et al.* 2006). Extensive flooding, water quality impairments, and environmental damage caused by older stormwater management techniques motivated a shift in management emphasis from simply conveying excess water to also protecting receiving waters (Jones *et al.* 2005). To better address stormwater-related issues, the US Congress amended the Clean Water Act (CWA) in 1990 and 1999 to include Phases I and II of the National Pollutant Discharge Elimination System (NPDES) stormwater control programs (USEPA 2005). The NPDES requires that stormwater be managed to the 'maximum extent practicable.' The Phase I program requires NPDES permits for large municipalities that have populations of 100,000 or more. The Phase II program extends permit requirements to smaller communities (<100,000 residents) that fall under the US Census Bureau's definition of an urbanized area, referred to as small *municipal separate storm sewer systems* (MS4s).

In communities operating under NPDES permits, the Oregon Department of Environmental Quality (ODEQ) requires that stormwater be managed under programs that address illicit discharge, construction site runoff, and post-construction management of stormwater in new developments and re-developed areas. Stormwater regulations apply to all construction sites disturbing more than one acre (0.4 ha), industrial sites, and MS4s. Entities managing municipal systems are responsible for establishing a stormwater management program that follows specific treatment requirements and also maintains compliance with Oregon's *total maximum daily load* (TMDL) process currently implemented by the ODEQ.

¹⁴ A shift to more uniform species assemblages and communities across the landscape, with a reduction of unique assemblages.

Smaller developments (i.e., with populations under 50,000) in Oregon that are not contiguous with urban areas demarcated by the US Census Bureau have not, historically, been required to apply for NPDES Phase II permits. Some of Oregon's cities with 20,000 or more residents that are currently not required to manage stormwater under the NPDES system include Albany, Grants Pass, McMinnville, Newberg, Redmond, Roseburg, and Woodburn (Oregon Environmental Council 2007). These cities address water quality impairments attributed to stormwater runoff under Oregon's TMDL process (Benninghoff 2009 pers. comm.¹⁵) which is implemented when water bodies no longer meet water quality standards developed for temperature, bacteria, and various pollutants (i.e., water bodies on the state 303d list).

The CWA also requires monitoring and regulation of all non-point source pollution (including stormwater) and ODEQ has recently dedicated additional resources to this end (ODEQ 2007). Due to the ubiquity of non-point pollution sources in urban and rural-residential developments, it will be years before Oregon realizes a comprehensive non-point pollution management program in developed areas (ODEQ 2007). Until then, the effects of non-point source pollution on aquatic ecosystems may rival or exceed those of point source pollution generated by developments (Line *et al.* 1997; Paul & Meyer 2001; USEPA 2002b).

The city of Portland provides a useful example of the challenges posed by stormwater management in Oregon cities that are experiencing rapid growth. Portland straddles five watersheds that collectively receive 80–100 billion gallons of precipitation annually. Twenty billion gallons of that precipitation are converted to stormwater runoff. By 2040, continuing development is predicted to increase this volume by roughly 10% (2.2 billion gallons; Vizzini 2007 pers. comm.¹⁶). Currently Portland's stormwater is conveyed through 568 miles of storm sewers and 861 miles of combined storm and sanitary sewers either to wastewater treatment facilities (see Section 3.0 of this report) or directly to the Willamette River and Columbia Slough. As Portland continues to grow, increasing stormwater volumes will strain the limits of this conveyance system. To ease the increasing demands on its stormwater management infrastructure, Portland is implementing initiatives that will encourage property owners to manage stormwater onsite rather than routing it to the sewer system. Initiatives include programs to disconnect stormflow from storm sewers (e.g., through downspout disconnections and installations of bioswales and rain gardens), to educate the public on stormwater-related issues and to facilitate the development of a local marketplace based on stormwater management¹⁷.

¹⁵ Benjamin Benninghoff, Oregon Department of Environmental Quality, Portland, Oregon, personal communication, March 28, 2009.

¹⁶ Vizzini D (Portland Bureau of Environmental Services) Presentation given at the Urban Ecology Research Consortium Symposium (Jan. 22, 2007), Portland State University.

¹⁷ For details visit <http://www.portlandonline.com/BES/index.cfm?c=44048>. Accessed November 22, 2010.

Key Findings–Stormwater:

- In communities operating under NPDES permits, ODEQ requires that stormwater point-sources be managed under programs that address illicit discharge, construction site runoff, and post-construction management of stormwater in new developments and re-developed areas.
- Entities managing municipal systems are responsible for establishing a stormwater management program that follows specific treatment requirements and also maintains compliance with Oregon's *total maximum daily load* (TMDL) process currently implemented by the ODEQ.
- In Oregon, some smaller cities that are not contiguous with urban areas demarcated by the US Census have not historically been required to apply for NPDES permits. These cities address water quality impairments attributed to stormwater point-sources under Oregon's TMDL process which is implemented when water bodies no longer meet water quality standards developed for temperature, bacteria, and various pollutants.
- It may be many years before Oregon realizes a comprehensive non-point pollution management program in developed areas.

Section 3.0: Wastewater

Urban wastewater reaches aquatic ecosystems from both point sources and non-point sources. Point source pollution comes from discrete conveyances (e.g., treated sewage drained through pipes), typically derived from domestic or industrial infrastructure. Non-point source pollution originates from diffuse sources such as stormwater runoff or improperly functioning septic systems. Wastewaters derived from rural-residential and urban areas contribute a variety of pollutants to aquatic ecosystems including personal care products, pharmaceuticals, fire retardants, excess nutrients and property maintenance chemicals (see Section 7.6 of this report). Industrial facilities produce comparatively large volumes of wastewater that often contain sewage and potentially toxic pollutants such as solvents and organochlorines (Anderson *et al.* 1996; Fuhrer *et al.* 1996; Wentz *et al.* 1998; Morace 2006). In stream or river reaches, where wastewater effluent is a substantial proportion of the flow, resulting water quality impairments can have negative consequences for aquatic biota (see Sections 7.6 and 8.0 of this report).

Section 3.1: Wastewater Management

In Oregon's urban areas, municipal wastewater treatment facilities process domestic¹⁸ and some types of industrial wastewater¹⁹. Some urban areas also have treatment facilities designed solely for industrial wastewater management. Treatment of domestic wastewater is a multistage process (Keller 1999; USEPA 2004b). *Primary treatment* involves screening large particulates from

¹⁸ Domestic wastewater includes wastewater from toilets, sinks, tubs, showers, and washing machines used in residences, businesses, and industries.

¹⁹ The following summary of how wastewater is managed in Oregon is based in part on personal communications with Ranei Nomura, Oregon Department of Environmental Quality, Portland, Oregon, May 20, 2010.

incoming wastewater then separating the remaining fine particulates by allowing them to settle out of suspension within quiet tanks or ponds. *Secondary treatment* uses aerobic and anaerobic bacteria to metabolize fine particulate and dissolved organic waste products that remain after primary treatment is completed. Secondary treatment also involves disinfecting water with chlorine, ozone, or ultraviolet light. Finally, *tertiary treatment* is used to remove or neutralize specific pollutants that pass through secondary treatment processes, such as excess nutrients, heavy metals, and organic compounds. Examples include air stripping to remove ammonia or chelating chemicals to remove nutrients and metals (Ragsdale 2007). Primary and secondary treatments are required in all of Oregon's municipal wastewater treatment facilities that discharge effluent into surface waters. In the northern Willamette River basin, several specialized facilities also use more expensive tertiary procedures. Industrial wastewater treatment processes vary depending on the type of industry producing the waste. For example, screening of large particulates (primary treatment) may not be required if they are not present in industrial wastewaters or disinfection may not be required when there is no human sewage in the wastewater.

The ODEQ currently administers two permit programs to regulate the discharge of treated wastewater to surface or ground waters (Table 3-1). The NPDES permits were established under the CWA and Water Pollution Control Facilities (WPCF) permits were established under Oregon's state waste discharge permit law. WPCF permits are used to regulate wastewater discharges that have the potential to negatively impact groundwater. The CWA requires state regulatory agencies with NPDES permitting authority (e.g., ODEQ) to ensure that wastewater treatment facilities comply with federal regulations adopted by USEPA and state water quality standards approved by the USEPA. Federal regulations typically set minimum treatment requirements that integrate the best available treatment technologies with economically feasible treatment options. State water quality standards set specific criteria that are necessary to protect beneficial uses of surface and ground waters²⁰. Beneficial uses include aquatic organisms and their habitats, recreational activities such as boating and swimming, and drinking water. The number of individual NPDES and WPCF permits issued by ODEQ (Table 3-1) indicate that surface and ground waters in Oregon could receive a substantial volume of treated wastewater. Many of these facilities currently operated by NPDES permit holders are discharging below their design capacity and can treat more wastewater. It is likely that discharge will increase as Oregon's human population continues to grow.

An NPDES permit limits the amount of monitored pollutants that can be discharged to surface waters by permit holders. This regulation applies to pollutants that have water quality standards set by the State of Oregon²¹. NPDES permits require that the concentration of a monitored pollutant present in wastewater effluent be reduced when the discharged effluent is likely to exceed state water quality standards for the pollutant of concern. If the pollutant concentration in wastewater effluent is at a level that meets state water quality standards, then further treatment to reduce the concentrations of that pollutant is not required. When a water body receiving wastewater effluent fails to meet state water quality standards, ODEQ performs a thorough analysis of point source wastewater and nonpoint source inputs as part of a TMDL analysis.

²⁰ For details on beneficial uses of surface waters in Oregon see: <http://www.deq.state.or.us/wq/standards/uses.htm>. Accessed May 20, 2010.

²¹ For up-to-date lists of regulated water pollutants see <http://www.deq.state.or.us/WQ/standards/standards.htm>. Accessed May 11, 2010.

TMDL analyses define how much of each monitored pollutant a river or stream can receive and still meet water quality standards²².

Table 3-1. Numbers of active discharge permits issued under the NPDES and WPCF programs administered by ODEQ*.

	Eastern Region	Northwest Region	Western Region	Statewide
NPDES INDIVIDUAL PERMITS	75	111	188	374
Major Domestic	9	17	23	49
Major Industrial	3	7	9	19
Major Stormwater	0	4	2	6
Minor Domestic	31	35	88	154
Minor Industrial	31	46	53	130
Minor Stormwater	1	2	13	16
WPCF INDIVIDUAL PERMITS	290	213	356	859
Domestic	74	15	53	142
Industrial	28	8	12	48
Stormwater	0	1	0	1
Onsite	188	189	291	668
ALL	365	324	544	1233

* Data were obtained at from ODEQ staff (Ranei Nomura, ODEQ, personal communication, May 5, 2010). For complete descriptions of permit categories, see the ODEQ Water Quality Permit Program website at <http://www.deq.state.or.us/wq/wqpermit/permitfaqs.htm>. Accessed on November 22, 2010..

ODEQ requires that NPDES permit holders adequately operate and maintain treatment facilities, monitor effluent discharges for concentrations of specified pollutants, and report findings to ODEQ on a regular basis. To help public agencies comply with their permit requirements, ODEQ uses its Clean Water State Revolving Loan Fund. The program for this fund was enacted in 1987 through amendments to the Federal Water Pollution Control Act. In Oregon, the fund provides low-cost loans for planning, designing, or constructing various water pollution control activities conducted by public agencies. Oregon's public agencies have made extensive use of this resource. ODEQ has loaned more than \$800 million since its program was established in 1990. For the 2010 fiscal year (July 1, 2009 – June 30, 2010) ODEQ made \$96 million available for projects aimed at improving water quality.

WPCF permits are designed to protect groundwater and comply with drinking water standards. For example, approximately one third of the WPCF permits issued by ODEQ regulate underground injection systems that involve the discharge of several types of wastewater below

²² For information on the analyses that have been done to date, visit ODEQ's website at <http://www.deq.state.or.us/WQ/TMDLs/basinlist.htm>. Accessed December 30, 2010.

ground. Underground injection systems are commonly used by cities or counties that do not operate municipal stormwater treatment facilities or by private businesses that are not connected to municipal sewer or stormwater systems. Because underground injection systems discharge wastewater effluent directly into the ground, they have the capacity to introduce pollutants to groundwater that can subsequently enter surface waters such as streams, rivers, or wetlands. Permit holders discharging treated wastewater effluent into below-ground structures such as sumps, drywells, or drain fields are required to monitor effluents and ensure that the wastewater discharge will not pollute groundwater²³.

Because of Oregon's permit regulations, wastewater treatment facilities in Oregon have generated a considerable amount of monitoring data on the quality of treated wastewater discharged into the environment. Published reports tend to be case studies of "best available technologies." For example, Ragsdale (2007) reported that advanced tertiary treatment reduced total suspended solids and total phosphorous at two Portland-area wastewater treatment facilities. Presently, not all of the monitoring data reported to ODEQ is stored in an easily accessible electronic format. Much of the data collected by NPDES permit holders operating in Oregon is held in regional ODEQ offices in hard copy format. In 2004 ODEQ developed an electronic discharge monitoring database system (known as DMS or Discharge Monitoring System) to track monitoring data from its permittees and has been able to consistently enter data from "major"²⁴ NPDES permittees into this database since 2006.

Many rural-residential communities throughout Oregon rely on septic systems for wastewater treatment (e.g., Mitchell & Harding 1996; ODEQ 2004; Hinkle *et al.* 2007), which are regulated by ODEQ issued WPCF permits. Similar to municipal treatment plants, septic systems use settling tanks and bacteria to separate and metabolize organic waste. The resulting liquids flow from septic tanks to subterranean drain fields which diffuse the remaining wastes through surrounding soils (USEPA 2004b). Where soil drainage is adequate, septic systems can be a safe, cost-effective means to treat wastewater. However, in locations where clay, sand, or shallow soils are common, water tables are high, or flooding occurs frequently, septic systems can discharge incompletely treated or untreated wastewater into local ground or surface waters (Keller 1999; USEPA 2004b).

Section 3.2: Wastewater-related Issues Important to Aquatic Ecosystems

Specific water quality impairments resulting directly from wastewater effluent cannot be identified because the primary literature on urban water quality (e.g., Sonoda *et al.* 2001; Carle *et al.* 2005; Atasoy *et al.* 2006) fails to discriminate the effects of wastewater effluent from those of other urban influences, such as stormwater runoff. Other considerations related to municipal wastewater treatment facilities and septic systems also make it very difficult to draw generalizations about the proportion of ecosystem impairment that is a direct result from wastewater.

²³ For additional information on the use and regulation of underground injection systems see <http://www.deq.state.or.us/wq/uic/uic.htm>. Accessed 21 May, 2010.

²⁴ 'Major' permittees are generally domestic treatment plants with design flows equal to or greater than 1 million gallons per day and larger industrial treatment plant in specified industrial categories.

Several Oregon cities route wastewater and stormwater to treatment facilities through the same set of pipes, referred to as a *combined sewer system*. Combined sewer systems can discharge untreated sewage and stormwater directly into rivers in what is termed a *combined sewer overflow* if stormwater exceeds the capacity of sewage lines and treatment facilities. The number of chemicals identified in wastewater treatment facility effluent and combined sewer overflows ranges up to 200 (Ritter *et al.* 2002) but the timing and magnitude of events that trigger combined sewer overflows make it difficult to generalize about the quantity of contaminants that actually reach receiving waters. The combined sewer system serving Portland, for example, discharged six billion gallons of combined sewer overflow into the Willamette River and Columbia Slough in 1990²⁵. Between 1980 and 2001, all but two of Oregon's 31 communities that operate combined sewer systems modified their facilities to reduce the frequency of combined sewer overflows²⁶. Two Oregon communities (Portland and Astoria) are still working with ODEQ to meet regulatory standards for combined sewer overflow events. To date, Portland reduced combined sewer overflow discharge to 2.1 billion gallons in 2006 and further system modifications are scheduled for completion in 2011; Astoria is scheduled to complete corrections by 2022 (ODEQ 2001a).

Biological contaminants and nutrients from poorly functioning septic systems can be a source of non-point pollution in rural-residential areas (Mitchell & Harding 1996; Conn *et al.* 2006; Swartz *et al.* 2006; Hinkle *et al.* 2007; Squillace & Moran 2007). As part of a cooperative effort to understand if and how groundwater resources are affected by septic tanks, the USEPA, US Geological Survey, ODEQ, and the Deschutes County Environmental Health Division developed the *La Pine National Demonstration Project* (La Pine Project). The La Pine Project assessed how septic systems are affecting groundwater within southern Deschutes County (Oregon) (Hinkle *et al.* 2005, 2007, 2009). Hinkle *et al.* (2005, 2007, 2009) tested both septic and groundwater samples, and found that groundwater samples contained high levels of septic-derived nitrate (NO₃; 1% of sites exceeded USEPA maximum contaminant level standards). In addition nine organic compounds including pesticides and pharmaceuticals were found at concentrations lower than those measured in septic water (Hinkle *et al.* 2005). Researchers working in other areas where large numbers of septic systems are used in Oregon (e.g., Mitchell & Harding 1996) and throughout the US (e.g., Conn *et al.* 2006; Swartz *et al.* 2006; Squillace & Moran 2007) have also documented septic derived substances moving into groundwater. These findings have raised concerns about how contaminated groundwater may be contributing to surface water quality impairments in the Deschutes and Little Deschutes Rivers (Williams *et al.* 2007).

It is difficult to draw generalizations about how septic systems affect surface water quality for two interrelated reasons. First, the infiltration rate of septic contaminants depends on local soil and hydraulic parameters; the same contaminants will not always pose equal risks to groundwater quality in different areas (Mitchell & Harding 1996; Hinkle *et al.* 2005). Second, it is unclear to what degree groundwater contaminants transfer to streams and rivers (Hinkle *et al.* 2005; Gaddis *et al.* 2006). Groundwater-surface water interactions are extremely dynamic and variable (e.g., Ellis *et al.* 2007; Keery *et al.* 2007), making it difficult to predict when septic

²⁵ Portland Bureau of Environmental Services, <http://www.portlandonline.com/bes/index.cfm?c=31030>. Accessed November 22, 2010.

²⁶ Regulations allow combined sewer overflows during 5-year return (winter), 10-year return (summer), or larger, storm events (Richard Santner, Oregon Department of Environmental Quality, Portland, Oregon, personal communication). Note that a 5-year return storm event, for example, has an exceedence probability in a given year of 20%, i.e., the average length of time between such events is five years (McCuen 1998).

leakage will pose a significant threat to aquatic biota within a given basin. It is likely though that high groundwater tables will facilitate septic contaminant percolation into surface waters (Kalbus *et al.* 2006). Using data from the 1990 US Census²⁷, IMST estimated the number of septic units in Oregon. Assuming the fraction of households with septic systems (29.3%) remained constant between 1990 and 2005; a rough estimate of the current number of household septic systems in Oregon is 456,000. Given the estimated and increasing number of septic systems operated in Oregon, water quality impairment stemming from malfunctioning septic systems is a legitimate concern warranting further investigation (Paul & Meyer 2001; USEPA 2002b; Mueller & Spahr 2006).

Key Findings-Wastewater:

- The numbers of individual NPDES permits issued by ODEQ indicate that surface water and groundwater in Oregon could receive a substantial volume of treated wastewater.
- The number of chemicals identified in wastewater treatment plant effluent and combined sewer overflows ranges up to 200, but the timing and magnitude of events that trigger combined sewer overflows make it difficult to draw generalizations about the quantity of contaminants that actually reach receiving waters.
- NPDES and WPCF permit regulations set limits on the amount of monitored pollutants that can be discharged to surface waters by permit holders. These regulations only apply to pollutants that have water quality standards set by the State of Oregon.
- Rural-residential communities throughout Oregon rely on septic systems for wastewater treatment. Biological contaminants and nutrients from poorly functioning septic systems can be a source of non-point pollution in rural-residential areas. It is likely that high groundwater tables and porous soils will facilitate septic contaminant percolation into surface waters. Given the estimated and increasing number of septic systems operated in Oregon, water quality impairment stemming from failed septic systems is a legitimate concern.

²⁷ Data available at http://factfinder.census.gov/home/saff/main.html?_lang=en. Accessed November 22, 2010. The 2000 and 2010 Census did not include wastewater information so only 1990 data could be included here.

Sections 4 to 8: Aquatic Ecosystem Processes Altered by Urban and Rural-residential Development

Sections 4 through 7 review how hydrology, physical habitat, water quality, and fish passage can be affected by urban and rural-residential areas. IMST reviews the factors and mechanisms responsible for changes to aquatic ecosystem condition in urban and rural-residential areas, and the nature, timing, and magnitude of these changes. In Section 8, IMST concludes Science Question 1 by summarizing the comprehensive effects of aquatic ecosystem alteration on aquatic communities, particularly algae, macroinvertebrate and fish assemblages, and salmonid populations.

Section 4.0: Hydrology

Hydrologic processes shape rivers and estuaries by moving water, wood, and sediments, which create and maintain conditions that support aquatic biota including salmonids (Poff & Ward 1989; Jay & Simenstad 1996; Poff *et al.* 1997; Borde *et al.* 2003; Konrad & Booth 2005; Beechie *et al.* 2006; Poff *et al.* 2006a). Within any watershed, several hydrologic pathways route precipitation toward runoff, groundwater, or the atmosphere. Partitioning of precipitation among these hydrologic pathways, in part, shapes the local hydrologic regime which for rivers is typically characterized by the source, magnitude, timing, frequency, duration, seasonality, and rates of change of high and low flows (Poff *et al.* 1997).

Within the Pacific Northwest, regional differences in climate and geology exert significant controls over the natural hydrologic regimes of streams, rivers, and estuaries (Figure 4-1; Poff 1996; Benke & Cushing 2005; Beechie *et al.* 2006; Sanborn & Bledsoe 2006). For example, watersheds west of the Cascades receive 75–80% of their precipitation from large winter fronts that generate peak flows from November to March (Redmond & Koch 1991; Good 1999, 2000; Colbert & McManus 2003; Kentula & DeWitt 2003). Many streams and rivers in central and eastern Oregon are heavily influenced by snow melt or short duration, but intense, summer thunderstorms. Consequently, peak flows east of the Oregon Cascade Mountains can occur during the winter, spring, or summer months (Sanborn & Bledsoe 2006). Throughout Oregon, the timings of winter peak and of summer low flows correlate strongly with local geology (Tague & Grant 2004).

Hydrologic pathways that are altered by urban and rural-residential development lead to changes in the natural hydrologic regimes (e.g., Poff *et al.* 2006a) which can have profound consequences for aquatic ecosystems and the salmonids that depend on them (Konrad & Booth 2005; Walsh *et al.* 2005b). Across the US, broad physiographic regions or ecoregions show different hydrologic responses to development (Konrad & Booth 2005; Poff *et al.* 2006a). These different responses highlight the importance of incorporating a landscape context (e.g., climate, geology, and land use history) when evaluating either the effects of development along Oregon's rivers and streams or the potential for rehabilitating hydrologic regimes in these systems (Konrad & Booth 2005). For example, within the Willamette River basin (western Oregon) the hydrologic response to urbanization and associated water quality impacts occur within the context of river basin-scale hydrologic regulation resulting from the operation of numerous dams (Gregory *et al.* 2007).

Climate also plays a major role in watershed hydrology and climate change has the potential to intensify or reverse the hydrologic effects of development (Claessens *et al.* 2006). General

circulation model projections consistently predict both warmer temperatures and changes in seasonal precipitation patterns in the Pacific Northwest during the 21st century. Warmer winter temperatures are projected to reduce snow pack levels, which would result in higher winter streamflows. Precipitation is projected to increase during winters, further augmenting winter streamflow levels. In summer periods, however, precipitation is projected to decrease, leading to lower streamflows (Payne *et al.* 2004, Markoff & Cullen 2008). Therefore, climate change may influence the type and timing of precipitation and have important effects on future snow packs and streamflows (Hamlet & Lettenmaier 1999; Leung & Wigmosta 1999; Miles *et al.* 2000; Mote *et al.* 2003; Payne *et al.* 2004; ISAB 2007a) and salmonids (Battin *et al.* 2007) throughout the Pacific Northwest.

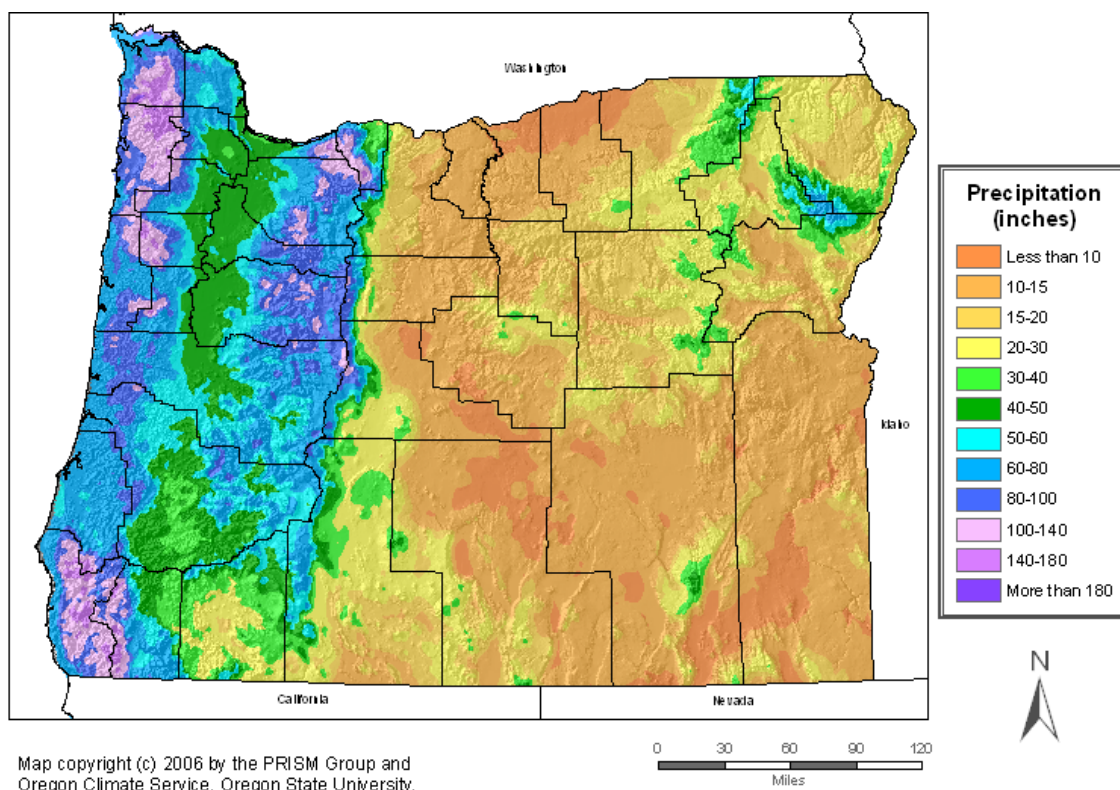


Figure 4-1. The average annual precipitation for Oregon calculated from 1971-2000. Source is the Oregon Climate Service, <http://www.ocs.oregonstate.edu/index.html>. Accessed on April 22, 2009.

Section 4.1: Factors and Mechanisms that Alter Hydrology in Urban and Rural-residential Areas

This section emphasizes changes in the timing and magnitude of streamflow, the linkage of streamflow to other physical and biological processes, and the importance of these processes in shaping salmonid habitats. Hydrologic interactions with channel conditions are discussed in Section 5.1 on physical habitat.

SECTION 4.11: IMPERVIOUS SURFACES

Within undeveloped lands, infiltration rates and water storage capacity are strongly influenced by soil characteristics; soil characteristics are determined by parent materials, climate, topography, biota and age (Gerrard 1981). The extent and distribution of soil infiltration capacity depend on how the mineral component of soils are leached by precipitation, soil particle size and heterogeneity (i.e., the relative composition of sand, silt and clay), and differ among regions with varying physical and biological characteristics (Hillel 1980). In intact forests and grasslands, rainfall not intercepted by vegetation infiltrates soils, after infiltration, soil water is mostly either withdrawn by vegetation or released groundwater.

One hallmark of development is the increased imperviousness of watersheds to precipitation. Construction of impervious surfaces, including pavement (e.g., roads, sidewalks, parking lots), roofs, and compacted soils, decreases infiltration and increases surface runoff directly routed to surface waters (Figure 4-2; Dunne & Leopold 1978; Paul & Meyer 2001; Booth 2000; Konrad & Booth 2005; Walsh *et al.* 2005a). The hydrologic effects include both faster stream response times and higher peak flow events, which are referred to as the “flashiness” of the stream. However, the fraction of impervious area that is created by development within a watershed and the hydrologic consequences of the impervious surface are context-dependent. Some studies conducted in the Pacific Northwest and elsewhere suggest that the degree of flashiness may be a function of watershed scale. Smaller watersheds tend to show flashier responses than larger watersheds. In some cases larger urbanized watersheds do not show changes in streamflow characteristics at all, presumably because of storage or flood control reservoirs and other landscape modifications that might increase water storage in the basin (Clark 1999; Chang 2007; Brown *et al.* 2009).

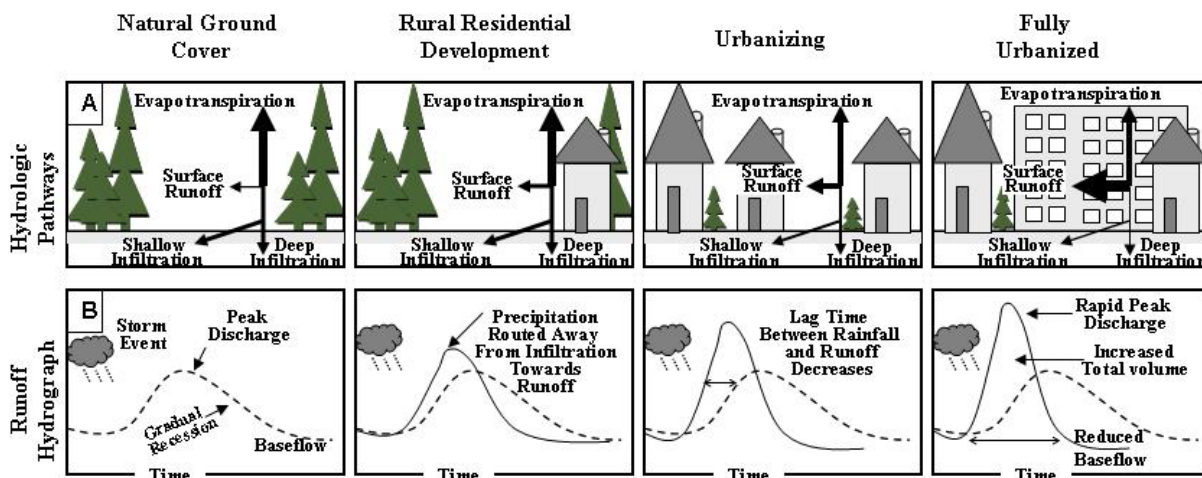


Figure 4-2. Hypothetical changes in precipitation routing among hydrologic pathways with increasing imperviousness area in watersheds undergoing urbanization. Arrow thickness represents the proportion of water taking each respective route but the relative proportions may change depending on climate, forest type, or other site specific factors. B. Comparative differences in the timing and magnitude of surface runoff from a single storm event and dry season base flows in undeveloped, developing, and developed watersheds. Dashed lines represent the predevelopment hydrograph while solid lines show changes in the hydrograph at progressive stages of development. Figure is modified from USEPA (1993) and Arnold & Gibbons (1996).

Because developed landscapes contain significant quantities of impervious surfaces, various measures of impervious area are commonly used to characterize the extent and intensity of development (Table 1-1) and to guide land use planning (Schueler 1994; Arnold & Gibbons 1996; Lee & Heaney 2003; Walsh *et al.* 2005a). In the Pacific Northwest, estimates of watershed-scale *total impervious area* (TIA) range from <5% in undeveloped watersheds (e.g., Reinelt *et al.* 1998) to 54% in the Columbia Slough watershed (City of Portland 2005). Site-level TIA associated with particular developed land uses (e.g., residential, commercial, industrial) can vary considerably depending on parcel dimensions and spatial arrangement of construction within parcels (e.g., Stone 2004; Stone & Bullen 2006). As a result, residential zoning and land use regulations, typically applied to individual parcels, can have significant control over watershed- and regional-scale imperviousness (See Section 9.2 of this report).

The connectivity between impervious surfaces and streams has important implications for the hydrology of rivers and estuaries in developed watersheds (Lee *et al.* 2006; Konrad & Booth 2005; Walsh *et al.* 2005a). With respect to aquatic ecosystem condition, the distinction between TIA and *effective impervious area* (EIA, the fraction of TIA with direct hydrologic connections to streams) has demonstrated importance (Hatt *et al.* 2004; Taylor *et al.* 2004; Walsh 2004; Newall & Walsh 2005). Gutters, drains, and stormwater sewers effectively bypass the infiltration capacity and time delays in natural soil systems and reduce the time required for surface runoff to enter streams, rivers and estuaries.

SECTION 4.12: CHANGES IN VEGETATION, INTERCEPTION, AND EVAPOTRANSPIRATION

Interception of precipitation by vegetation plays a significant role in local hydrologic processes because the complex surfaces of plants delay precipitation from reaching the ground and route a significant fraction of precipitation to evapotranspiration (Figure 4-2). In forested areas, interception can range from 10 to 50% of annual precipitation depending on forest structure (e.g., tree species, tree size, epiphyte load, deciduous vs. evergreen foliage) and the timing, form, and intensity of precipitation (Waring & Running 1998). Removal of native upland and riparian vegetation during rural-residential and urban construction, followed by replanting with smaller, often non-native, species may affect annual interception and subsequently evapotranspiration. In a long-term study of 51 watersheds in the eastern US, Dow & DeWalle (2000) documented decreased watershed-scale evapotranspiration where urban land cover increased more than 10% over the study period. Overall, reducing interception and evapotranspiration rates increase the proportion of water traveling through watersheds as surface runoff (Sanders 1986) and the rate at which precipitation is converted to runoff. This reduction in interception and evapotranspiration rates intensifies the damaging effects of impervious surfaces and stormwater (Dow & DeWalle 2000; Xiao & McPherson 2002).

American Forests (2001) conducted a landscape analysis of the Willamette and lower Columbia River regions to determine how forest cover had changed between 1972 and 2000. Findings from this analysis relevant to Oregon's aquatic ecosystems were:

- Across the region, light tree canopy cover increased from <20% to 75% of the land area while areas of heavy (natural) tree cover declined by 56%;
- Tree cover within urban areas (i.e., Albany, Beaverton, Corvallis, Eugene, Portland, Salem, Tualatin, and Wilsonville) declined from 21% to 12%; and
- Tree loss resulted in an estimated 7.2 billion gallon increase of stormwater production per peak storm event (2-year return, 24-hour storm event²⁸).

In a similar analysis of the Puget Sound region (Washington), American Forests (1998) found that tree cover loss and associated increases in impervious surfaces between 1972 and 1996 resulted in a 35% increase in stormwater production. Throughout the Pacific Northwest, loss of forest cover on low-elevation private forestland is expected to accelerate in the near future where these properties are converted to rural-residential developments (Stein *et al.* 2005).

SECTION 4.13: WATER WITHDRAWALS AND TRANSFERS

Rural-residential and urban development of landscapes diverts water resources toward residential, recreational, landscaping irrigation, commercial, and industrial uses, including waste processing facilities. Data collected at the county level (Table 4-1) illustrate these water withdrawal/transfer imbalances across Oregon. For example, both Multnomah and Washington counties have small within-county water withdrawals given their population sizes, while adjacent Clackamas County has comparatively large water withdrawals.

²⁸ A 2-year return, 24-hour storm event is a storm of 24 hour duration with such magnitude (i.e. rainfall depth) that it has an exceedence probability in a given year of 50%; i.e., the average length of time between such events is two years (McCuen 1998).

The magnitude of water removal from and transfer into watersheds for human uses can influence streamflow in rural-residential and urban waterways, particularly during drought periods. For example, several major cities in the Portland (Oregon) metropolitan area import a significant portion of their municipal water supplies from the Bull Run watershed, which lies outside the Willamette River basin. This diversion of water significantly reduces streamflow in both the Bull Run and Sandy Rivers with negative consequences for several threatened salmonid species (NMFS 2006b). Similarly, the Tualatin River water is used heavily for municipal purposes during the summer months but several major cities within this watershed also import water from the Bull Run watershed. Such water transfers may in part explain increases in both summer flows and annual runoff to precipitation ratios observed as urban development in the Tualatin watershed increased between 1976 and 2000 (Chang 2007). Additionally, increases of water transfer into urban systems can result in negative consequences for the urban streams receiving that additional water, as discussed in more detail in the Section 7.0 of this report.

Municipal and other water withdrawals from rivers that feed Oregon's estuaries produce poorly understood hydrologic effects. Reduced freshwater flows have the potential to alter sediment transport and delta morphology (Jay & Simenstad 1996). When summer freshwater flows into estuaries fall below historical levels, the freshwater/saltwater interface at the tidal front travels farther upriver (Good 2000). The ecological consequences of such changes for salmonids and their predators, competitors, and prey are unknown.

For the period 2000 to 2050, Houston *et al.* (2003) projected freshwater withdrawal increases in Oregon of 65% for public and domestic uses and 59% for industrial and commercial uses. Increasing demands on water resources imposed by population growth and increasing development pose significant threats to aquatic biota. Climate changes that reduce water quantity and increase summer stream temperatures will likely exacerbate these issues in many parts of Oregon (ISAB 2007a).

Table 4-1. Water withdrawn for public supply in Oregon counties in 2000. Public supply refers to water withdrawn by public and private water suppliers that provide water to at least 25 people or have a minimum of 15 connections (for domestic, commercial, industrial, or thermoelectric-power use).

Oregon County	Population in 2000 ¹	Surface Water Mgal/Day ¹	Ground Water Mgal/Day ¹	Total Mgal/Day ¹
Baker	16,741	2.50	0.63	3.13
Benton	78,153	5.71	0.12	5.83
Clackamas [*]	338,391	167.22	6.53	173.75
Clatsop	35,630	9.13	0.00	9.13
Columbia	43,560	3.46	2.67	6.13
Coos	62,779	5.08	2.48	7.56
Crook	19,182	0.00	1.45	1.45
Curry	21,137	2.02	0.81	2.83
Deschutes	115,367	6.54	15.17	21.71
Douglas	100,399	5.82	0.00	5.82
Gilliam	1,915	0.00	0.29	0.29
Grant	7,935	2.27	0.60	2.87
Harney	7,609	0.00	2.06	2.06
Hood River	20,411	2.91	0.24	3.15
Jackson [*]	181,269	38.71	0.46	39.17
Jefferson	19,009	2.69	0.40	3.09
Josephine	75,726	5.29	0.18	5.47
Klamath	63,775	0.02	9.96	9.98
Lake	7,422	0.33	1.85	2.18
Lane [*]	322,959	35.30	22.39	57.69
Lincoln	44,479	4.32	0.47	4.79
Linn	103,069	11.88	0.48	12.63
Malheur	31,615	2.96	2.60	5.56
Marion [*]	284,834	76.89	8.61	85.50
Morrow	10,995	2.31	2.76	5.07
Multnomah ^{\$}	660,486	1.10	9.42	10.52
Polk	62,380	2.78	1.84	4.62
Sherman	1,934	0.00	0.43	0.43
Tillamook	24,262	2.32	1.41	3.73
Umatilla [*]	70,548	14.92	10.56	25.48
Union	24,530	0.00	3.72	3.72
Wallowa	7,226	0.82	0.14	0.96
Wasco	23,791	1.28	0.90	2.18
Washington ^{\$}	445,341	23.14	3.03	26.17
Wheeler	1,547	0.13	0.23	0.36
Yamhill	84,992	7.31	3.56	10.87

¹ Population data are from US Census Bureau (2003). Water withdrawal data are from US Geological Survey District Water Use Program (Oregon) http://or.water.usgs.gov/projs_dir/or007/or007.html (accessed February 15, 2007).

^{*} The top five counties (Clackamas, Jackson, Lane, Marion, and Umatilla) for combined surface water and groundwater withdrawals.

^{\$} Counties (Multnomah and Washington) with low total water withdrawals for their population size.

SECTION 4.14: CHANNELIZATION AND FLOOD CONTROL

River and estuary channelization is frequently done to “improve” lands for agricultural, residential, and industrial land uses (IMST 2002b). Oregon’s larger rivers and estuaries have been channelized to make them navigable for shipping and transportation (Cortwright *et al.* 1987; Benner & Sedell 1997; NMFS 2007). Channel modifications can protect buildings and roads from localized flooding and erosion, accelerate storm runoff conveyance, and facilitate draining floodplain wetlands. Some channelization projects have been designed specifically to minimize the space occupied by a channel and maximize the amount of real estate available for development (reviewed in Riley 1998). As a result, habitat for aquatic organisms is severely degraded or lost completely. Channelization also has large implications for stream geomorphology and physical habitat modification, as discussed in Section 5.2 of this report.

Actions commonly taken to channelize rivers and estuaries include:

- Removing wood, sandbars, and rocks that form channel constrictions;
- Installing bank stabilization structures such as revetments, riprap, and seawalls for flood and erosion control;
- Disconnecting sloughs, alcoves, side-channels, and chutes through the installation of dikes and levees that confine streamflow to a single channel; and
- Straightening meanders to speed water transport through the channel.

The Willamette River, which flows through Oregon’s largest urban areas, has been highly modified with numerous bank stabilization structures. Gregory *et al.* (2002b) estimated that approximately 26.5% of Willamette River mainstem miles have revetments on one or both banks. Between Eugene and Newberg (Oregon) many of these structures function to prevent soil erosion from agricultural and forested lands. However, both the number of revetments and the percentage protecting developed lands increase considerably along the river miles between Newberg and the Willamette’s confluence with the Columbia River (Gregory *et al.* 2002b; Table 4-2). Although rivers were originally channelized and revetments placed primarily for navigation and the protection of agricultural lands, revetments are increasingly associated with developed lands.

Table 4-2. Percentage of revetments along the Willamette River (Oregon) designed to protect municipal and rural-residential land uses.

Land Use Adjacent to Bank Stabilizing Structures (Revetments) ¹			
Willamette River Reach	Number of Revetments	Percent Municipal or Rural-residential	Percent Agricultural, Forestry, Bare Land
Columbia Confluence to Newberg	138	64	26
Newberg to Albany	117	26	74
Albany to Eugene	113	22	78

¹ Data from Gregory *et al.* (2002b).

Channelization is often undertaken as a means of local flood control, but it can contribute to increased flood levels by reducing the amount of overbank storage (i.e., floodplain storage; Woltemade 1994). Storm runoffs accumulate rapidly and produce higher and more frequent flood peaks compared to stream channels that remain connected to wetlands and side channels capable of absorbing floodwaters. For example, the lower 15 miles of Johnson Creek, a tributary of the Willamette River in Portland were heavily channelized during the 1930s to control flooding (Johnson Creek Watershed Council 2003). Impervious surfaces constructed since the 1940s have increased the peak runoff volumes in Johnson Creek by 30%. As a result of increased impervious area Johnson Creek regularly overflows its banks and floods several residential areas in Portland (Johnson Creek Watershed Council 2003; Chang 2007).

SECTION 4.15: WETLAND LOSS AND ALTERATION

Many of Oregon's urban and rural-residential areas are located along lowland aquatic ecosystems that serve important hydrologic and ecological functions (reviewed by IMST 2002b). Development in these ecosystems isolates, fragments, and eliminates wetlands (Holland *et al.* 1995; Thom *et al.* 2001) reducing their ability to collect and redistribute flood flows, recharge groundwater aquifers, store water for slower release, and provide rearing and overwintering habitat for fish and wildlife, particularly coho and Chinook salmon.

Gregory *et al.* (2002a) estimated that almost 95% of Willamette Valley wetlands present in 1851 were converted to other land cover types by 1990. Willamette Valley wetlands continue to disappear (Daggett *et al.* 1998; Bernert *et al.* 1999) despite environmental regulations established during the 1970's to prevent future wetland loss. Net wetland loss occurring between 1981 and 1994 in the Willamette Valley was estimated to be 10.6 sq. mi. (Bernert *et al.* 1999) and 9.9 sq. mi. (Daggett *et al.* 1998) in independent assessments. For this period, the fraction of lost wetland area converted to urban development was estimated at six percent (0.64 sq. mi.; Bernert *et al.* 1999) or 9% (0.90 sq. mi.; Daggett *et al.* 1998), while the wetland area converted to rural-residential development was estimated at 15% (1.49 sq. mi.; Daggett *et al.* 1998). The balance of lost wetland area was converted to agriculture or other land uses (Daggett *et al.* 1998).

Within the Portland metropolitan UGB, 40% of wetlands existing in 1982 were lost by 1992 (Kentula *et al.* 2004). Similarly, 6% of wetlands remaining within the Portland metropolitan UGB in 1992 were lost by 1998. Kentula *et al.* (2004) also found that impervious surfaces increased the magnitude of water level fluctuations and reduced floodwater retention capacity of small (< 5 acres) wetlands within the Portland metropolitan UGB. Hydrologic alterations also shifted the majority of remaining wetlands from riverine (within a river channel) to types with consistently deep water levels (ponds) that are atypical of the Portland region (Kentula *et al.* 2004). Consequently, altered hydrology and ecological function in remaining wetlands reduce their capacity to support native biota (Kentula *et al.* 2004).

As rural-residential and urban development increases, the quantity and quality of wetland habitat available to salmonids can be expected to deteriorate. Across Oregon, urban and rural-residential land cover and development rates are highest in western Oregon, particularly the Willamette Valley (Azuma *et al.* 2002, 2004, 2009). It is reasonable to assume that wetland loss will also result from human population growth and development in other areas of Oregon. Rural-residential and urban development have been identified as a threat to agricultural lands (Meyer & Turner 1992; Ramankutty *et al.* 2002; Burcher & Benfield 2006) which, while highly altered,

still have the capacity to provide some wetland habitat. Conversion of agricultural lands to urban and rural-residential uses may reduce the potential for rehabilitating these lands to benefit water quality, fish, and wildlife.

Section 4.2: Nature, Magnitude, and Timing of Hydrologic Changes

Studies conducted worldwide document the hydrologic changes that accompany development activities such as those described above (Konrad & Booth 2005; Walsh *et al.* 2005a; Chin 2006). In this section, IMST summarizes research on the nature, magnitude, and timing of these changes, with emphasis on available information from the Pacific Northwest. Changes to hydrologic routing commonly observed in developed landscapes include changes in *surface runoff* (quantity and timing), as well as changes in *groundwater recharge* and *stream base flow*.

SECTION 4.21: CHANGES IN SURFACE RUNOFF AND INFILTRATION CAPACITY

Landscape alterations that accompany development drive fundamental changes in the timing, magnitude, and nature of surface runoff and flood events in developed watersheds (Konrad & Booth 2005; Walsh *et al.* 2005b). Impervious surfaces increase the runoff component of local hydrologic regimes (Figure 4-2). Using data from 40 monitoring sites across the US, Schueler (1994) found that the fraction of precipitation volume routed to surface runoff (i.e., the runoff coefficient) increased directly with an increase in watershed imperviousness. This finding was later corroborated in a review by Shuster *et al.* (2005), who reported the median runoff coefficient increased to above 80% on highway monitoring sites where imperviousness was at a maximum. Vegetation alteration and surface leveling prior to construction and wetland loss exacerbate the reduced infiltration capacity of the land surface. Wetland loss also reduces water storage potential and increases the rate at which water moves through watersheds (Thom *et al.* 2001). Ultimately, water that may otherwise take hours, days, or weeks to move through the soil reservoir flows rapidly over hardened surfaces and enters streams in short, intense pulses.

Increased imperviousness and associated drainage structures alter the timing of surface runoff in two ways:

- *lag time* between peak precipitation and peak runoff (i.e., the time it takes rainfall to become runoff) decreases, causing streamflow to rise more rapidly during storms; and
- duration of *peak runoff* decreases resulting in more rapid recession of streamflow after storms.

Because increased imperviousness decreases infiltration, a larger proportion of precipitation becomes runoff. As a result, the magnitude of surface runoff changes as both the volume of individual runoff events and frequency of bank-full flow events increase compared to undeveloped watersheds. Because many Pacific Northwest soils have naturally high infiltration capacity, the increase in surface runoff due to development is often greater than increases observed in other geographic regions (Booth & Jackson 1997). The net result of changes in streamflow timing and magnitude are surface runoffs that are larger in volume, that enter stream channels more rapidly, and that reach bank-full volumes more frequently compared to undeveloped watersheds (Figure 4-2). Streams flowing through developed watersheds are often described as 'flashy' (Booth *et al.* 2004; Konrad & Booth 2005), a characteristic associated with

increased streamflow, reduced infiltration within the stream channel and flood plain, and a larger fraction of precipitation exiting watersheds as surface runoff (Paul & Meyer 2001; see Section 5.1 of this report for additional information on the consequences of such changes). Diminished infiltration can also convert wetlands that are consistently flooded into seasonal wetlands that dry out during the summer (Thom *et al.* 2001). Because the majority of runoff exiting a watershed is simply shifted from infiltration to surface flow, development may not affect annual (or longer term) streamflow (Konrad & Booth 2002). In watersheds where vegetation alterations affect evapotranspiration patterns, or where water transfers into or out of the watershed occur, annual streamflow may either increase or decrease depending on the watershed-scale water balance (Konrad & Booth 2005; Chang 2007).

While the reach-scale effects of impervious surfaces can be severe, contributions of increased surface runoff to watershed-scale stream degradation also depend on watershed size, development intensity, development placement within the watershed, and the efficiency with which runoff is routed from impervious surfaces to streams (Walsh *et al.* 2005a). Burges *et al.* (1998) found that surface runoff ranged from 12 to 30% of annual precipitation in a forested Puget Sound (Washington) lowland watershed, while surface runoff ranged from 44 to 48% of annual precipitation in a nearby watershed with 30% TIA. Using reach-scale data, Poff *et al.* (2006a) found that Pacific Northwest watersheds with 15 to 30% urban land cover had decreased high flow duration, increased flashiness, and annual flood peaks 22 to 84% greater than those in undisturbed watersheds. The magnitude of change in hydrologic behavior was larger in the Pacific Northwest than other physiographic regions in the US (Poff *et al.* 2006a). For example, a flashiness index (Sanborn & Bledsoe 2006) comparison between undisturbed and urbanized watersheds showed that developed Pacific Northwest watersheds were more flashy than comparable watersheds in southeastern and southwestern US regions (Poff *et al.* 2006a).

Information on the net effect of development outside UGBs on watershed hydrology is sparse. However, simply removing forest cover during rural-residential development is known to decrease evapotranspiration and increase streamflow (Grimmond & Oke 1986; Dow & DeWalle 2000; Claessens *et al.* 2006). Data from rural-residential developments in Washington indicate that reduced infiltration rates resulting from lawns and pastures on hobby farms (as reported by Booth *et al.* 2002), and suburban landscaping (Konrad 2003) are sufficient to significantly increase streamflow. Similarly, Burns *et al.* (2005) compared the hydrologic response of suburban watersheds in northern New York with 6.2% and 11.1% TIA to an undeveloped watershed and documented respective increases in peak discharge of 42% and 300%.

The nature of surface runoff changes can depend on the spatial patterning of rainfall relative to the location of development and naturally impervious landscape features within a river basin (Smith *et al.* 2005). For example, Hollis (1975) used data from several studies to examine the general relationship between flood size and impervious surface area effects. Hollis (1975) found that the total volume of stormwater increased with increasing flood size and recurrence interval, but the fraction of stormwater generated by impervious surfaces decreased. Hollis's (1975) observations could be explained by soils becoming saturated during prolonged storms and exhibiting hydrologic behavior similar to impervious surfaces (Chin 2006). Increased surface runoff from saturated soils may manifest more rapidly on thin soils associated with lawns and other landscaping in developed areas (Konrad 2003; Alberti *et al.* 2007). Research results from

the *Baltimore Ecosystem Study*²⁹ (Maryland) indicate that, in addition to TIA, both antecedent soil moisture and land surface heterogeneity explain variation in the fraction of precipitation that becomes runoff during discrete storm events (Smith *et al.* 2005) and support conclusions drawn by Hollis (1975). However, Smith *et al.* (2005) also demonstrated that an efficient drainage network structure can diminish the hydrologic effects of various landscape characteristics, thus highlighting the importance of hydrologic connectivity between impervious surfaces and streams noted by Walsh *et al.* (2005a).

The general hydrologic response of arid and semi-arid aquatic ecosystems to rural-residential and urban development is understudied relative to more mesic regions and warrants increased investigation. In arid regions, natural hydrologic regimes exhibit high spatial and temporal variability in the form, timing, and magnitude of precipitation. As a result, their hydrologic response to precipitation can be episodic and varied (Gray 2004a). Thus, hydrologic change associated with urbanization in more arid climates is likely to be difficult to predict compared to mesic climates (Chin & Gregory 2001). In undeveloped semi-arid climates, vegetation is relatively sparse, soils are shallow, and overland flow can be the predominant route of precipitation to streams (NRC 2002). In semi-arid areas where a high percentage of precipitation naturally exits watersheds as surface flow, the proportional increase in surface flow resulting from development may be small compared to what would occur in mesic areas experiencing similar levels of development (Konrad & Booth 2005). However, increasing urbanization around streams that are already naturally flashy may cause significant increases in peak flow (Chin & Gregory 2001).

Conversely, inter-watershed water transfers and development of municipal and industrial wastewater treatment facilities that discharge effluent directly into river channels can convert naturally ephemeral or intermittent streams into perennial systems. In many arid regions, treated wastewater effluent may make up the majority of flow during periods of low precipitation leading to serious water quality issues (Brooks *et al.* 2006). This can also occur in relatively humid regions when cities discharge wastewater effluent to small streams during periods of low flow.

SECTION 4.22: CHANGES IN GROUNDWATER RECHARGE AND BASE FLOW

The base flow (or non-storm flow) of many Pacific Northwest rivers is derived primarily from groundwater. Leopold (1968) and Harbor (1994) predicted that increased imperviousness should decrease base flow because an increased proportion of precipitation is routed away from groundwater towards surface runoff (Figure 4-2). Channelization and associated wetland drainage can also reduce water tables and affect surface flows (Thom *et al.* 2001). Observations of groundwater and low streamflow reductions associated with increasing urbanization support Leopold's (1968) and Harbor's (1994) predictions (Klein 1979; Simmons & Reynolds 1982; Ku *et al.* 1992; Barringer *et al.* 1994; Rose & Peters 2001; Fitzpatrick *et al.* 2005).

However, other studies of streamflow responses to urban development present conflicting findings, indicating that urbanization does not affect baseflows consistently. Poff *et al.* (2006a) found reduced minimum streamflows at the reach-scale associated with greater than 15 to 30%

²⁹ The Baltimore Ecosystem Study is one of 26 LTER (Long-term Ecological Research) programs established by the National Science Foundation. See <http://www.beslter.org/>. Accessed on October 12, 2010.

urban land cover in the Pacific Northwest but also documented regional differences in the minimum flow response to increasing urban land cover. Konrad & Booth (2002) found that while winter (wet season) base flows in western Washington streams declined in response to urban development, minimum annual 7-day streamflow showed inconsistent results. They concluded that poor infiltration reduces shallower subsurface water stores that feed winter base flows more than deeper groundwater recharge that supports summer base flows. Others have documented that leaky water and sewer lines (e.g., Lerner 1986, 1990, 2002) can actually increase base flows in urban developments. In urban areas, irrigation used for landscape plants, gardens, and lawns provides additional input to the water balance (Grimmond & Oke 1986) and perhaps contributes to groundwater and to base flow. Brandes *et al.* (2005) analyzed 25 years of streamflow data from six watersheds undergoing urbanization and four reference watersheds in the Delaware River basin (Pennsylvania and New Jersey) and found that watershed geology, location of development within a watershed, and anthropogenic water transfers were more useful in explaining base flow patterns than impervious surface estimates. This inconsistency in response of base flow patterns to increasing impervious surface area (or other indices of development) may be scale-dependent (Brandes *et al.* 2005), but clearly more investigation is needed in this area.

Key Findings: Hydrology

- Impervious surfaces and soil compaction decrease infiltration and increase surface runoff that is directly routed to surface waters. Residential zoning and land use regulations can affect watershed- and regional-scale imperviousness.
- Removing native vegetation during construction, and replacing with smaller, non-native species, can reduce interception and evapotranspiration rates and increase the fraction of stormwater that reaches impervious surfaces.
- Many of Oregon's developments reduce the ability of wetlands to collect and redistribute flood flows, recharge groundwater aquifers, store water for slower release, and provide rearing and overwintering habitat for fish.
- Changes to hydrologic routing commonly observed in developed landscapes include changes in surface runoff as well as changes in groundwater recharge and stream base flow. The hydrologic effects include faster stream response times to precipitation, higher and shorter peak flow events, and potentially reduced base flows.
- Contributions of increased surface runoff to watershed-scale stream degradation depends on watershed size, development intensity, development placement within the watershed, and the efficiency with which runoff is routed from impervious surfaces to streams.
- The general hydrologic response of arid and semi-arid aquatic ecosystems to development is understudied relative to more mesic regions and warrants increased investigation.
- Climate change will play a major role in watershed hydrology and has the potential to intensify the hydrologic effects of development. Increasing demands on water resources imposed by human population growth pose significant threats to aquatic biota. Climate changes that reduce water quantity and increase summer stream temperatures will likely exacerbate these issues in many parts of Oregon.

Section 5.0: *Riparian and Aquatic Physical Habitat*

Physical habitat influences the abundance and distribution of organisms. In aquatic environments, physical habitat variables include channel substrate size, channel complexity and cover, riparian vegetation structure, and the degree and type of channel-riparian interaction. Physical processes that work in combination to create a wide variety of riparian and physical habitats include (Montgomery & Buffington 1998):

- Timing and magnitude of streamflow,
- Wood and sediment supply, transport and deposition,
- Stream bank and bed erosion, and
- Channel migration.

The spatial and temporal complexity of physical habitat conditions formed by these processes provide elements critical to salmonid persistence. Rural-residential and urban development can significantly modify the processes that link aquatic ecosystems to the surrounding landscapes and lead to physical habitat deterioration by altering hydrology, riparian vegetation, and river channel structure.

Section 5.1: *Factors and Mechanisms that Alter Physical Habitat in Developed Areas*

This section emphasizes the effects of development on stream condition from a physical structure (i.e., channel stability, substrate, large wood) viewpoint. This means that reach-scale characteristics of physical habitat will often be emphasized within the text. Whenever possible, the IMST attempts to put these reach-scale changes and their local effects on salmonids in the context of the larger terrestrial and aquatic landscape.

SECTION 5.11: HYDROLOGIC MECHANISMS

Hydrologic processes interconnect strongly with processes that shape physical habitat in streams, with each continuously responding to changes in the other. Unless stream channels are naturally constrained by bedrock or some similar means, they will make dynamic adjustments in response to changes in flow regime and sediment supply imposed by development (Dunne & Leopold 1978; Booth & Henshaw 2001; Bledsoe 2002). Streamflows carrying sediment and large wood interact with channel beds and banks to shape channel form (Schumm 1971; Dunne & Leopold 1978; Beschta 1985; Beschta & Platts 1986). Stream type and mode of sediment transport play significant roles in determining stream response to altered hydrology. Consequently, hydrologic changes are important forces driving channel readjustment in developed landscapes (Walsh *et al.* 2005b). Changes in channel morphology can in turn become mechanisms for further local hydraulic change (Figure 2-1; Poff *et al.* 2006a).

Rural-residential and urban developments alter watershed hydrology in several ways (see Section 4.1 of this report). Impervious surfaces in developed watersheds create surface runoff that is larger in volume, enters stream channels more rapidly, and reaches bank-full volumes more frequently compared to undeveloped watersheds. Stormwater detention facilities cannot fully

mitigate changes in surface runoff volumes; to reduce peak discharges, they extend the duration of flows at or just below bank full volumes, which can have the effect of inducing channel erosion (Booth & Jackson 1997; Brown & Caraco 2001; Bledsoe 2002). Intentional channelization, whether for commerce, bank stabilization or flood control purposes, increases channel capacity, stream power, and the potential for channel erosion and flooding downstream (Simon & Rinaldi 2006). Reduced channel and physical habitat complexity are almost universal consequences of increased flashiness and channelization that accompany development (Walsh *et al.* 2005b; Poff *et al.* 2006a).

Riparian vegetation adds channel complexity and bank stability directly from roots or indirectly as large wood when trees fall into streams. Hydrologic changes alter interactions between streams and adjacent riparian areas in ways that can remove established riparian vegetation and/or prevent re-colonization or regeneration of riparian plants in riparian zones (Booth 1991). Many riparian plant species are adapted to the hydrologic patterns of flood and drought in undeveloped landscapes. Thus, hydrologic alterations can affect the extent and structure of riparian vegetation communities that establish and grow along riverbanks in developed watersheds. For example, black cottonwood (*Populus trichocarpa*) and willow (*Salix* spp.) seed germination and establishment requires bare mineral soil deposition and slow flow recession following floods (Dykaar & Wigington 2000; Amlin & Rood 2002; Karrenberg *et al.* 2002). Fierke & Kauffman (2005) found that new black cottonwood regeneration along the Willamette River (Oregon) was limited to low areas experiences annual flooding and scouring.

SECTION 5.12: UPLAND DISTURBANCE

Construction activities mechanically disturb large quantities of soil, thereby increasing erosion and watershed sediment yields (i.e., hillslope erosion; Wolman & Schick 1967; Keller 1999; Paul & Meyer 2001). Areas disturbed by construction can have soil erosion rates 2 to 40,000 times greater than pre-disturbance rates and annually produce ~80 million tons of sediment that enter US waters (Harbor 1999). Compacted soils have reduced infiltration rates and subsequent increased surface runoff across these soils leads to increased erosion (Leopold 1968; Keller 1999). Road building is another key source of sediment delivered to waterways (Waters 1995).

The increase in erosion rates caused by construction and the amount of sediment transported to surface waters depends on watershed size and development intensity. Based on 100 published studies conducted worldwide over the past 50 years, sediment production in urbanizing watersheds tends to increase 2- to 10-fold (Chin 2006). In western Washington, Nelson & Booth (2002) documented nearly a two-fold increase in watershed-scale sediment erosion in the urbanizing Issaquah River basin (144 km²) while only 0.3% of the watershed area was under construction. During the later stages of urbanization when construction activities subside, impervious surfaces, compacted surfaces, landscaping, and stormflow detention and treatment can limit sediment mobilization and reduce sediment delivery rates (Paul & Meyer 2001; Chin 2006; Poff *et al.* 2006a).

SECTION 5.13: RIPARIAN VEGETATION REMOVAL

Salmonid and steelhead population declines in the Pacific Northwest have been attributed, in part, to lost and reduced riparian functions (Beschta 1997). Riparian ecosystems are zones of dynamic interaction between aquatic and terrestrial ecosystems that exert significant influence over streamflow dynamics, instream habitat structure, and interactions among stream channels and off-channel habitat (Gregory *et al.* 1991). As a result, aquatic ecosystem condition is strongly associated with the condition of riparian vegetation (Stoddard *et al.* 2005; Paulsen *et al.* 2008). For example, intact riparian areas facilitate channel floodplain interactions that provide important off-channel rearing areas for juvenile salmonids (Sommer *et al.* 2001).

The spatial distribution and structural characteristics of riparian vegetation vary within and across physiographic regions in Oregon. Riparian vegetation is influenced by site and watershed characteristics including stream size, channel gradient, plant community composition, disturbance regimes, interactions among channels, and subsurface water movement (Anderson *et al.* 2004; Everest & Reeves 2007). Consequently, the minimum area of undisturbed riparian vegetation required to preserve riparian ecological function is variable depending on site and watershed characteristics. In general, riparian vegetation contributes to:

- Stream bank formation,
- Stream bank stabilization,
- Large wood contributions,
- Channel complexity maintenance,
- Stream temperature modification from shade,
- Organic litter contributions that fuel aquatic food webs,
- Terrestrial food sources (e.g., leaf litter, insects) for aquatic organisms, and
- Sediment and nutrient filtration from stream inputs.

Riparian areas are ecologically critical for maintaining healthy aquatic ecosystems, but they also tend to be valuable real estate properties and development often results in the restructuring or total loss of riparian vegetation (Ozawa & Yeakley 2007). In the greater Portland metropolitan region, approximately 51% of riparian buffers along the 775 stream miles within the UGB were developed by 1998 (Ozawa & Yeakley 2004). In portions of the Columbia River basin, important to ESA-listed salmonid populations, riparian areas affected by urban and agricultural development were significantly narrower than riparian areas in forest or shrub/grassland landscapes (i.e., 30 m vs. 70 m median width, respectively; Fullerton *et al.* 2006).

Riparian deforestation is often identified as an important driver of the aquatic ecosystem response to development (Figure 5-1; e.g., May *et al.* 1997; Booth 2005; Hook & Yeakley 2005; McBride & Booth 2005). May *et al.* (1997) estimated that when TIA exceeded 40%, approximately 40% of riparian buffers no longer provided ecological benefits to streams draining small Puget Sound (Washington) watersheds (Figure 5-1). McBride & Booth (2005) attributed heterogeneous physical habitat condition in moderately urbanized areas, in part, to intactness of the local riparian buffer. Similarly, Hook & Yeakley (2005) concluded that near-stream riparian buffers were important for maintaining water quality in Johnson Creek (Portland, Oregon). However, riparian habitat loss covaries with several other factors making it difficult to isolate

riparian alterations as a direct cause of aquatic ecosystem impairment (Ourso & Frenzel 2003; Walsh *et al.* 2005b). For example, road crossings, that interrupt otherwise intact riparian buffers, create points where stormwater can bypass routing and detention facilities and flow directly into streams, thus degrading physical habitat (McBride & Booth 2005). The importance of riparian area continuity along streams has been documented by others (reviewed by Fullerton *et al.* 2006), but the minimal lengths and widths of contiguous riparian buffers required to maintain aquatic communities will likely vary with environmental characteristics such as stream size, topography, parent materials and climate factors.

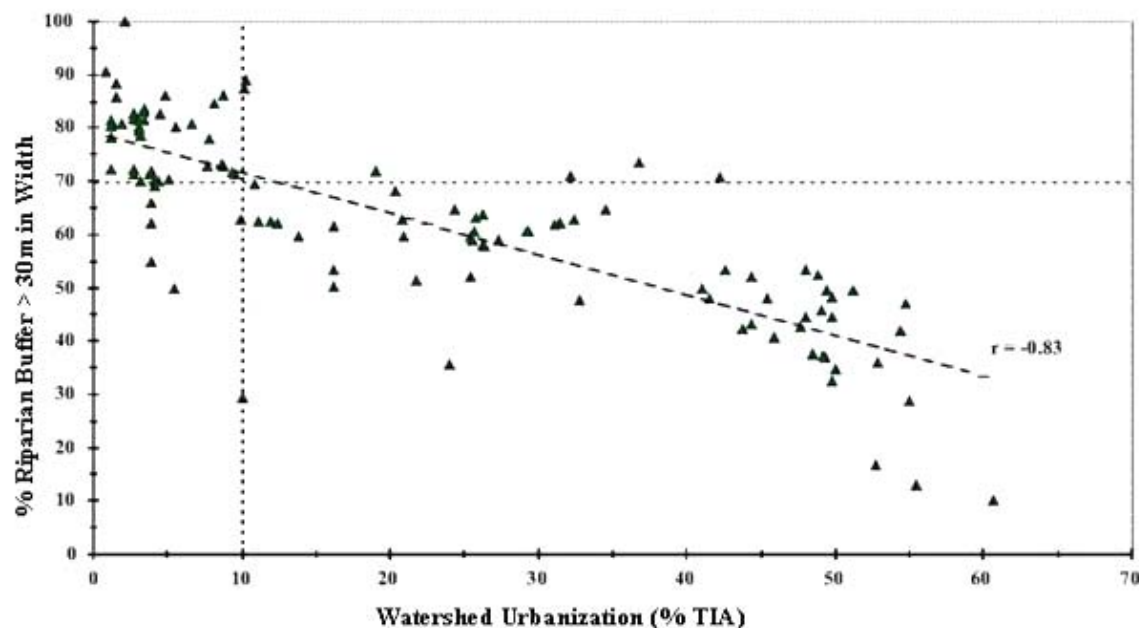


Figure 5-1. The relationship between riparian buffer width and total impervious area (TIA) in small Puget Sound lowland streams. Horizontal and vertical dashed lines highlight sites where 70% or more of riparian buffers were more than 30m wide and watershed TIA was less than 10%. The original authors used horizontal and vertical dashed lines to demarcate both the size of riparian buffers required by sensitive area ordinances in Puget Sound (Washington) and the level of watershed urbanization where most riparian buffer widths fell below this minimum size (i.e., 100–150 ft or 30 m). IMST feels that the vertical and horizontal lines should not be interpreted as implicit thresholds, as the existence of thresholds in this context has been the subject of considerable debate. Figure reproduced from May *et al.* (1997) with permission from the Center of Watershed Protection (Ellicott City, Maryland).

SECTION 5.14: DIRECT INSTREAM MODIFICATION OF PHYSICAL HABITAT

Restructuring river and stream channels can increase stream power, decrease channel complexity, and increase within-channel erosion. These modifications intensify the effects of altered hydrology and sediment delivery from upland sources (Jacobson *et al.* 2001). The removal of riparian vegetation and large wood, channelization, bridge construction, and channel dredging each result in significant channel erosion (Booth 1991; Waters 1995; Keller 1999).

Section 4.0 described several channel modifications that straighten rivers and confine flow to a single channel. This section reviews several additional ways that human activity directly alters aquatic physical habitat in developed areas including:

- Overwater structures,
- Dredging,
- Wood removal,
- Aggregate removal, and
- Estuary diking and filling.

Overwater Structures – Pilings, wharfs, and bridges can affect aquatic physical habitat structure. The level of disturbance created by these structures ranges from small changes associated with bridges constructed in rural-residential and urban settings to major habitat alterations associated with harbor development in large cities (Fresh *et al.* 2005; NMFS 2007). In small rivers, overwater structures can narrow channel width and create catch points for sediment and debris that increase resistance to flow. In estuaries, shade from overwater structures negatively affects eelgrass (*Zostera* spp.) beds and substrate disturbance disrupts epibenthic fauna³⁰ (Glasby 1999; Shafer 1999; Simenstad *et al.* 1999; Haas *et al.* 2002; Southard *et al.* 2006; NMFS 2007). Individual structures in rivers and estuaries can also modify riparian vegetation, stream bank structure, channel substrate composition, channel depth, and light contrasts (Simenstad *et al.* 1999; Carrasquero 2001; Kemp *et al.* 2005; Southard *et al.* 2006; NMFS 2007; Toft *et al.* 2007) in ways that may:

- Influence salmonid migratory behavior,
- Alter salmonid prey distributions and densities,
- Increase habitat overlap between salmonids and their predators, and/or
- Increase salmonid predator concentrations.

While individual structures may not cause significant impacts, the cumulative effects of numerous structures on river and estuary habitats can lead to fish assemblage changes (Jennings 1999; Simenstad *et al.* 1999; Carrasquero 2001; Haas *et al.* 2002; Southard *et al.* 2006).

Dredging – Material removal from channels, or dredging, is commonly done to increase channel depth or width and to reduce stream meandering (USFWS 2006; Wallick *et al.* 2009). Because urban areas are centers for commerce, major rivers are often dredged to develop and maintain navigable channels and ports for ships. In Oregon, the lower section of the Willamette River between Swan and Sauvie Islands, referred to as “Portland Harbor,” is routinely dredged to maintain a shipping channel 300 ft wide and 40 ft deep (Weston 1998). Dredging also occurs in eleven urban estuaries along the Oregon coast. Eight of the estuaries are designated by DLCD for shallow draft development³¹ and dredged when the bottom is at 22 ft or less while the remaining three estuaries are designated for deep draft development³² and dredged to maintain depths

³⁰ Organisms living on the surface of bottom sediments in streams, rivers, or estuaries.

³¹ Nehalem Bay, Tillamook Bay, Depoe Bay, Siuslaw River, Umpqua River, Coquille River, Rogue River, Chetco River.

³² Columbia River, Yaquina Bay, Coos Bay.

greater than 22 ft (Cortright *et al.* 1987; Good 1999). Dredging, together with altered flow and sediment regimes, has reduced the area of the Columbia River estuary by 20% (Fresh *et al.* 2005). Dredging can significantly increase channel capacity and gradient, triggering unintended changes in channel morphology throughout a watershed or river basin including upstream degradation (head cutting), downstream aggradation (filling), and bank instabilities throughout, as well as disconnecting the main channel from adjacent wetlands (Good 1999; Simon & Rinaldi 2006).

Large Wood Removal – During the 19th and 20th centuries, large wood was frequently removed, or snagged, from streams and estuaries to increase river navigability. In smaller channels, large wood and log jams were removed because they were perceived as barriers to migrating salmonids. These actions greatly reduced wood abundance in channels and wetlands of large rivers and estuaries of the Pacific Northwest (Collins *et al.* 2002; Wing & Skaugset 2002; Hood 2007). Snagging is still a common practice in rural-residential and urban areas (Larsen *et al.* 2004). Bridges and culverts tend to accumulate wood that must be removed to maintain these structures. Where streams run through parks and near homes, wood is likely to be removed to reduce the risk of accidents during recreational use of waterways. The removal of wood from channels in streams and estuaries decreases hydraulic roughness (i.e., channel structures that slow water velocity) which can increase flow velocities, stream power, peak discharge, and channel incision (Good 1999; Simon & Rinaldi 2006). Reductions of in-channel wood also decrease the abundance and surface area of pool habitat and channel refuges in streams and estuaries (Montgomery *et al.* 1995; Collins *et al.* 2002) required by salmonids (e.g., Montgomery *et al.* 1999).

Aggregate Removal – Sand and gravel are mined from waterways and floodplains for use in building and road construction, pipeline bedding, drainage areas (e.g., leach fields in septic systems), industrial applications (e.g., glass manufacturing, foundry operations, abrasives), and landscaping. River substrates are often preferred over upland sources because moving water physically abrades sediments removing weaker materials and naturally grades, sorts, and rounds the sediments. Demand for mined sand and gravel is often greatest where there is a convenient, economical mode of transport such as barges, and/or nearby markets, like urban areas (OWRRI 1995). Within Oregon, the majority of permitted aggregate mining operations are in the Willamette Valley followed by the Umpqua River basin. According to Kondolf (1994) mining river sediments can alter aquatic physical habitat by affecting:

- Habitat quantity (e.g., when river flow captures floodplain mining pits),
- Channel geometry (e.g., width-to-depth ratios),
- Channel gradient (when channels are deepened and straightened),
- Channel stability, and
- Channel incision rates.

Estuary Diking and Filling – Historical conversion of estuarine swamps and salt marshes to agricultural land through diking and draining accounts for most of the habitat loss in Oregon estuaries (Good 1999; IMST 2002b). However, recent and historical development of Oregon tidelands has also contributed to estuarine habitat loss (Cortwright *et al.* 1987; Good 1999,

2000). For example, urban and industrial development has reduced tidally influenced habitat in Coos Bay (Oregon) (Hoffnagle & Olson 1974; Borde *et al.* 2003), but the quantity of salmonid habitat lost to development coast-wide has not been estimated.

Estuaries are composed of hydrologically interconnected habitats that are especially important to juvenile salmonids (Cortwright *et al.* 1987; Good 1999, 2000). The fry of some salmonid species (particularly chum and Chinook) rear in the upper reaches of estuaries for several weeks and follow tidal fluctuations as they acclimate to the increased salinity of seawater (Groot & Margolis 1991; Thorpe 1994; Bottom *et al.* 2005). As juvenile salmonids mature, they move into deeper, more saline channels, mudflats, and eelgrass bed habitats in the lower estuary prior to migrating into the ocean (Groot & Margolis 1991; Thorpe 1994; Miller & Sadro 2003; Hosack *et al.* 2006). Filling tidelands in upper and lower estuary reaches to create urban and rural-residential properties, as well as armoring estuary shorelines to protect these properties, can destroy, fragment, and disconnect habitats necessary for the rearing, acclimation, and smoltification of salmonid juveniles (Good 1999, 2000; Borde *et al.* 2003; Bottom *et al.* 2005; Fresh *et al.* 2005; Rice 2006).

Section 5.2: Nature, Timing, and Magnitude of Physical Habitat Changes

Changes in salmonid habitat resulting from development range from minor effects to near complete loss of useable fish habitat. This section evaluates the nature, timing, and magnitude of change in aquatic habitat with respect to:

- Channel morphology,
- Channel and streambank stability,
- Large wood and other organic inputs, and
- Substrate siltation and coarsening.

The general direction of change in physical habitat caused by development is frequently documented, but the timing and magnitude of such changes are not well documented. The latter are more difficult to summarize because river channels continue to evolve in response to changing water flows, wood delivery and transport, and sediment loads, and may take decades to reach dynamic equilibrium (e.g., Henshaw & Booth 2000). When development and redevelopment activities are continuously occurring, stream channels may never reach a dynamic equilibrium (Chin 2006). The magnitude of change is partially dependent on the spatial pattern and sequence of landscape alterations as well as the underlying parent geology and hydrologic regime (Montgomery 1999; Henshaw & Booth 2000; Poff *et al.* 2006a, b). The direction and magnitude of change are also dependent on when measurements are made relative to the stage of development; streams may be aggrading or eroding, contracting or enlarging, and the magnitude of change may be increasing or decreasing over time (Chin 2006).

SECTION 5.21: CHANNEL MORPHOLOGY

Fluctuations in stream channel morphology reflect a dynamic equilibrium between erosion caused by bank full flows and more gradual delivery and stabilization of materials from upstream and terrestrial upslope areas. As a result, stream velocity, stream gradient, and the amount, type and size of sediment transported by the stream exert strong controls over stream channel dimensions (Bledsoe & Watson 2001b). Long-term changes in sediment budgets and flow regimes associated with development cause channel morphology to change until it arrives at a dynamic equilibrium structure that accommodates the modified conditions (Dunne & Leopold 1978; Booth 1990; Montgomery & Buffington 1997; Paul & Meyer 2001; Chin 2006). Morphological alterations typically observed include changes in channel depth, incision rate, width, sinuosity, number and size of point bars, and patterning of pools and riffles (Paul & Meyer 2001).

Periods of active construction disturb large quantities of soil and increase watershed sediment yields (Wolman & Schick 1967; Keller 1999; Paul & Meyer 2001). Channels undergo a series of erosional stages when sediment loads are delivered to streams experiencing increased flood frequencies, water volumes, and velocities, leading to an imbalance between sediment delivery and the stream's ability to transport sediment (Figure 5-2; reviewed in Paul & Meyer 2001; Chin 2006; Simon & Rinaldi 2006). An accumulation of excess sediment in river channels (aggradation) can reduce channel capacity and increase the number of flood events (Paul & Meyer 2001) that deposit sediment on stream banks and elevate bank heights above predisturbance levels. The magnitude of channel aggradation depends on watershed and stream size relative to the amount and location of development.

During the latter stages of urbanization, impervious surfaces cover soils and greatly reduce the supply of channel forming materials such as sediment and large wood. As hydrologic alterations stemming from increased impervious area intensify, stream channels begin to enlarge and destabilize riparian vegetation (May *et al.* 1997; Bledsoe & Watson 2001a; Bledsoe 2002; Konrad & Booth 2002; Simon & Rinaldi 2006). Stream channels typically enlarge through lateral erosion, bed erosion, and/or less often overbank deposition (Henshaw & Booth 2000). Depending on channel slope and geology, enlarging channels may incise up to several meters below their predevelopment bed levels (Booth 1990), or split into multiple (braided) channels winding between new gravel bars formed by increased sediment inputs (Booth & Henshaw 2001). Channel incision and widening limit interaction between streams and their floodplains, minimizing the potential benefits of riparian areas (Roy *et al.* 2006).

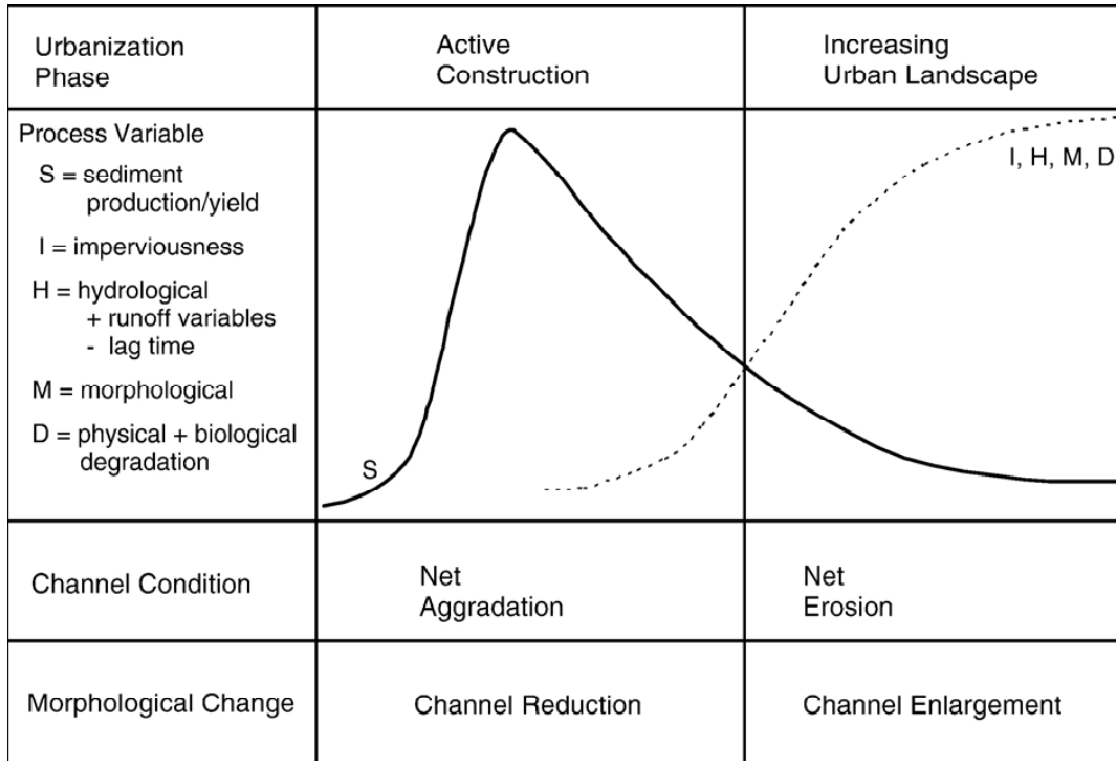


Figure 5-2. General phases of urbanization with associated process changes, channel conditions, and morphological adjustments. Curves are approximated to indicate trends but could exhibit considerable variation among variables depending on the hydrologic and geomorphic context within any particular watershed. Figure reproduced from Chin (2006) with permission from Elsevier B.V.

As channels enlarge, they become capable of containing stormflow events of increasingly larger volumes and the likelihood of floods overtopping stream banks decreases (Klein 1979; Booth 1991). Additional bank-full discharge events are a major source of channel erosion and can result in higher and steeper streambanks that are susceptible to mass failure events (Bledsoe & Watson 2001b; Simon & Rinaldi 2006). At this point, the majority of sediment transported by the stream comes from within-channel sources as opposed to hillslope erosion (Trimble 1997). Accelerated channel erosion can become a significant sediment source for downstream reaches (Simon & Rinaldi 2006) and can propagate the effects of development beyond the reach scale. Depending on the extent and intensity of development, it is not uncommon for channel alterations to migrate up and downstream of rural-residential and urban areas (Booth & Henshaw 2001; Simon & Rinaldi 2006).

Several studies have documented the magnitude of change in channel morphology with increasing development. Chin (2006) found that channels tended to enlarge to 2 to 3 times their original size in humid and temperate environments. Hammer (1972) studied 78 small watersheds near Philadelphia (Pennsylvania) and found that channel cross sectional area increased by a factor of 2.2 for impervious area associated with houses, and ranged up to a factor of 6.8 for impervious area associated with commercial buildings, row and apartment houses, factories,

airport runways, shopping centers, and parking lots. In King County (Washington), Booth & Reinelt (1994) observed an average channel widening of 0.6m (17%) along urban streams where native vegetation was significantly altered or removed.

As stream channels deepen and widen in response to development, other physical characteristics change as well. The pattern of aquatic habitat diversity (pools, riffles, runs) also changes as streams undergo the hydrologic and physical changes that accompany development (Paul & Meyer 2001). Booth & Reinelt (1994) evaluated fish habitat quality in two urbanizing Puget Sound (Washington) watersheds by measuring pool to riffle ratios and channel roughness and diversity; then rated habitat quality as excellent, good or degraded. Habitats rated as excellent or good generally were located in watersheds with less than 6% impervious area. Degraded habitat was first encountered in watersheds with 8% or more effective impervious area and most degraded habitat occurred in reaches with 10% or greater effective impervious area.

Overall, the rate and magnitude of changes in channel morphology depend on development type, watershed size, topographical relief, climate, geology, sediment sources, and land use history (Booth & Henshaw 2001; Paul & Meyer 2001). The sediment production (construction) phase can last months to years depending on the type of development. If and when sediment yields decline, channels may require several decades to adjust before they reach a dynamic equilibrium (Chin 2006). This assumes, of course, that the channel form was in dynamic equilibrium prior to development. Bledsoe & Watson (2001a) suggested that stream channels draining smaller, more permeable watersheds are more likely to exhibit a greater erosive response to development. However, because the primary controlling factors are significant watershed-scale characteristics and processes that operate over variable timeframes, the relationship between the rate and magnitude of channel response and measures of development (e.g., TIA) is highly variable (Henshaw & Booth 2000; Anderson *et al.* 2004) and varies along the length of the effected channel (Gregory *et al.* 1992; Paul & Meyer 2001).

Limited research in arid environments suggests that river responses to urbanization occur rapidly over short distances but are variable (Chin 2006). Many ephemeral streams naturally carry large suspended sediment loads and experience high spatial and temporal variability in precipitation inputs. Consequently, channel morphology may undergo rapid and less predictable responses to development that may also be more isolated because of the localized nature of rainfall (Chin 2006). Additional research is required to better understand the effects of development on aquatic physical habitat in Oregon's arid environments.

SECTION 5.21: CHANNEL AND STREAMBANK STABILITY

Streambank and channel stability are subjective concepts because their nature is inherently dynamic. Streambank erosion is a natural part of dynamic stream channels, particularly alluvial channels that are not constrained by bedrock and are composed of material transported from upstream reaches. For the purposes of this section, 'stable' streambanks and channels are those that show no continued, directional change in response to development.

The relationship between development and channel stability is complex and depends on flow regime changes, riparian vegetation condition, floodplains, and the intrinsic stability of the stream type (Bledsoe & Watson 2001a, b; Sudduth & Meyer 2006). Riparian vegetation is an important stream bank stabilizer and contributes to the maintenance or re-establishment of

channel stability by reducing erosion along both stream banks and streambeds and by dissipating stream power, especially during floods (Everest & Reeves 2007). Channel widening is linked to channel instability and mass wasting events. Widening streams and rivers gradually undercut their banks until the overhanging material collapses into the channel, thereby causing additional increases in sediment delivery from within the channel (Neller 1988; Ritter *et al.* 1995). Once stream channel capacity increases to the point where increasingly large peak flows are fully contained, the former floodplain no longer functions to dissipate flood energy. In this situation, stream channels may become unstable (Booth & Jackson 1997; Simon & Rinaldi 2006).

Forest retention can play an important role in channel stability. Booth *et al.* (2002) reported that 65% forest retention was required to maintain 2-year discharges below the 10-year modeled discharge. Similarly, May *et al.* (1997) found that streambed erosion became evident when TIA ranged from 10-30% and that streambanks generally were unstable when TIA exceeded 30% in Puget Sound lowland watersheds. Booth & Jackson (1997) characterized a large number of stream channels in Washington and related channel stability to percent EIA and surface runoff in their respective watersheds (Figure 5-3). Nearly all channels classified as 'stable' were in watersheds with less than 10% EIA. Conversely, most channels classified as 'unstable' were in watersheds with more than 10% EIA. Channels also tended toward instability when actual 2-year discharge exceeded the 10-year discharge modeled for each stream. This discharge benchmark generally occurred when EIA exceeded 10%, and mirrors Bledsoe's (2002) finding that channel stability depends on the frequency of moderate flow events and the distribution of stream power along weak points in the channel. While these findings suggest TIA or EIA thresholds at which significant changes in channel characteristics occur, there is considerable debate on the existence and applicability of thresholds in these systems.

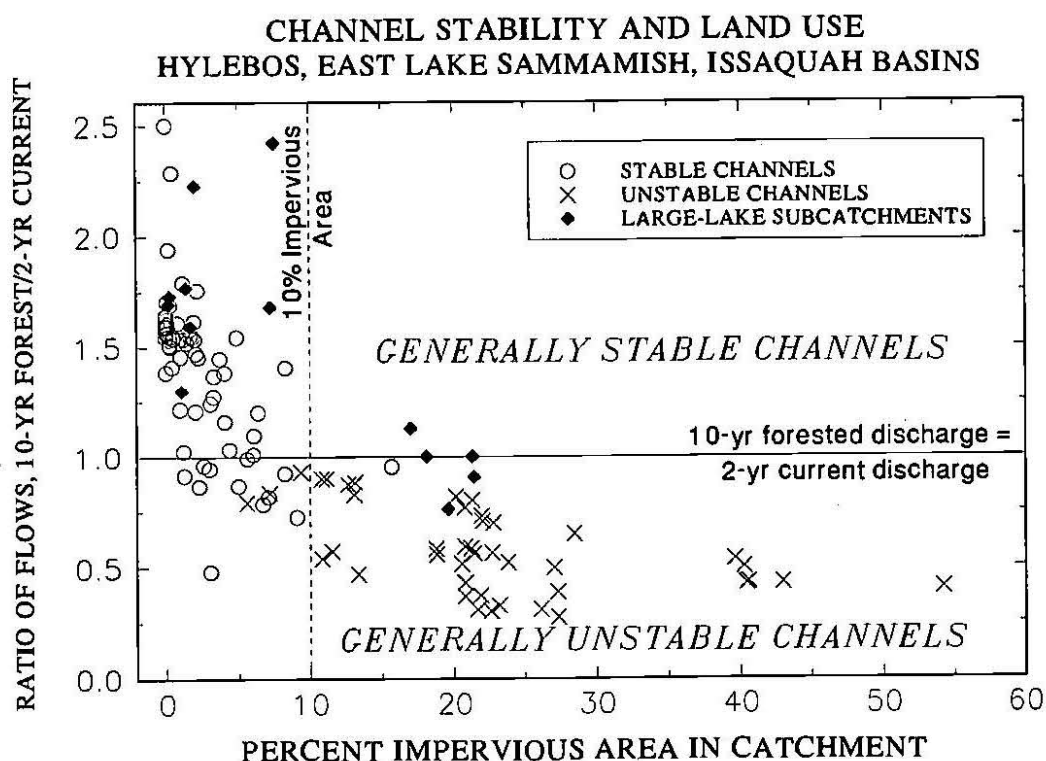


Figure 5-3. Channel stability as a function of surface runoff and EIA. These results are for basins dominated by glacial till soils and with no stormwater mitigation in the Hylebos, East Lake Sammamish, and Issaquah basins in western Washington. The vertical axis represents the ratio of modeled 10-year discharge for a forested basin to the current two-year discharge and gives an index of hydrologic change. When this ratio equals one (horizontal line), the 2-year current discharge equals the 10-year modeled discharge. Channel stability appears to consistently decline as this ratio decreases and EIA increases. IMST feels that the vertical and horizontal lines should not be interpreted as implicit thresholds, as the existence of thresholds in this context has been the subject of considerable debate. Reproduced from Booth & Jackson (1997) with permission from John Wiley and Sons.

SECTION 5.13: LARGE WOOD AND OTHER ORGANIC MATTER INPUTS

Large wood, particularly coniferous, is a key structural component influencing channel dynamics and aquatic habitat complexity in Pacific Northwest rivers, streams, and estuaries. The abundance, size, and abundance of individual large wood pieces influence the capacity of instream wood to form and maintain complex aquatic physical habitat. The importance of individual large wood pieces generally increases with increasing piece size (Bilby 1984; Bisson *et al.* 1987; Bilby & Ward 1989). Long, large diameter pieces trap additional materials, and change the streamflow dynamics and channel morphology (Collins *et al.* 2002). Many of the following functions served by large wood are of critical importance to the maintenance of salmonid habitat (Meehan *et al.* 1977; Bisson *et al.* 1987, 1988; Maser *et al.* 1988; Gregory *et al.* 1991; Hicks *et al.* 1991; Reeves *et al.* 1993; Montgomery *et al.* 1995, 1996; Beechie & Sibley 1997; Bilby & Bisson 1998; McIntosh *et al.* 2000):

- Dissipation of streamflow energy,

- Regulation of sediment storage and transport,
- Streambank protection and stabilization,
- Pool, riffle, and gravel bar formation, and
- Habitat diversity created by cover and channel complexity.

Large wood in streams, rivers, and estuaries comes from riparian and upslope areas. Large wood derived from riparian areas typically enters streams through windfall or stream bank erosion while more episodic landslides or debris torrents deliver wood from upslope sources. Periods of high water may subsequently transport large wood downstream from its entry point. Research conducted in western Oregon and Washington has shown that the majority of large wood (both number of pieces and volume) in a specific stream reach typically originates from windfall or erosion of local riparian forests (Keller & Swanson 1987; McDade *et al.* 1990; McGreer & Andrus 1992).

Several mechanisms associated with development reduce potential wood recruitment into streams. Past and current wood removal activities associated with development contribute to declines in large wood (May *et al.* 1997). As channels adjust to altered sediment budgets and flow regimes, increased stream power can flush remaining and newly recruited wood from the channel. Channel straightening and bank armoring reduces large wood recruitment by preventing rivers from laterally eroding their banks and floodplains (Collins *et al.* 2002). Levee construction, overbank deposition, and channel incision that isolate rivers from their floodplains also reduce the potential for rivers to recruit wood from forested floodplains (Roy *et al.* 2005). Finally, sources that resupply large wood to stream channels are reduced or lost as development encroaches on riparian forests (May *et al.* 1997; Finkenbine *et al.* 2000; Ourso & Frenzel 2003). Ultimately, the loss of large wood from river channels and estuaries decreases habitat complexity and intensifies the effects of channelization.

Wing & Skaugset (2002) found that land ownership and forest age were important predictors of large wood volume and size of structurally important logs in western Oregon streams. Their analysis indicated that rivers flowing through rural-residential developments contained significantly less wood than those in forested areas. In Puget Sound lowland streams, large wood abundance is 1 to 2 orders of magnitude below abundance estimates from the period prior to Euro-American settlement (Collins *et al.* 2002). In these streams, smaller, non-coniferous pieces that decay more rapidly dominate contemporary wood recruitment. Poor wood recruitment rates limit the number of 'key' logs large enough to initiate and stabilize wood jams (Collins *et al.* 2002). Finkenbine *et al.* (2000) found that British Columbia (Canada) streams flowing through landscapes with greater than 10% TIA contained minimal quantities of large wood unless substantial riparian buffer zones were present. Similarly, May *et al.* (1997) found that declines in both the size (>0.5m diameter) and quantity of large wood were correlated with increasing TIA (Figure 5-4). Parallel measures of salmonid rearing habitat (pool area, pool size, pool frequency) showed that the loss of large wood was associated with impairment of habitats important to salmonid populations (May *et al.* 1997). While the contribution of development to large wood reductions was not explicitly quantified, watersheds with TIA estimates below 5% and relatively intact upstream riparian buffers tended to contain the highest frequencies and volumes of large wood (May *et al.* 1997). This is consistent with conclusions drawn by Gregory *et al.* (1991) that intact, mature riparian forests are key to maintaining in-stream large wood.

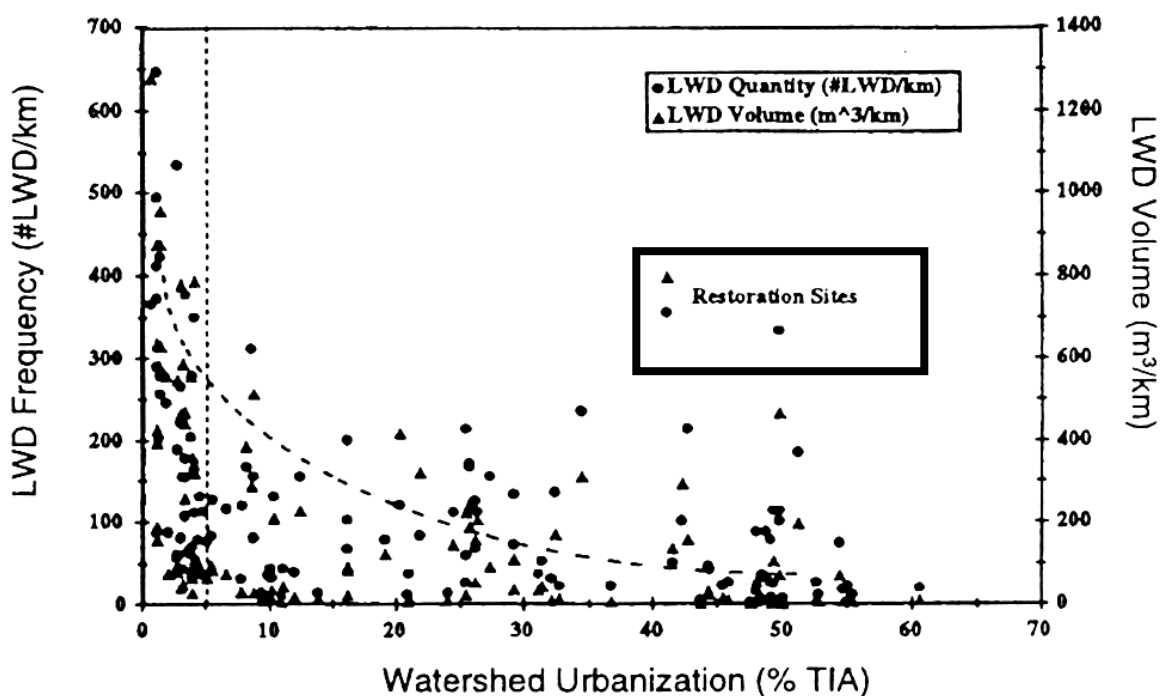


Figure 5-4. Large wood in streams in relation to total impervious area in the watershed. The vertical line highlights the finding that the amount of large wood (number of pieces or volume) in streams is highly variable even at low levels of TIA, but only sites with TIA below 5% had the highest large wood frequency. IMST feels that the vertical line should not be interpreted as an implicit threshold. May *et al.* (1997) attributed variation in large wood amounts to legacy effects of past timber harvest in some watersheds. Data points that fall within the inset box ($n=3$) received habitat restoration treatments. Figure reproduced from May *et al.* (1997) with permission from the Center for Watershed Protection (Ellicott City, Maryland).

SECTION 5.14: SUBSTRATE SILTATION AND COARSENING

Hydrologic and geomorphic processes that sort and deposit gravel and fine sediments maintain channel substrate structure. May *et al.* (1997) estimated that ecologically significant increases in the percent of fine sediment may occur when upstream TIA exceeded 20%. Excess fine sediments can cover streambeds, fill spaces between coarse substrate materials, and increase substrate embeddedness (Wolman & Schick 1967; Klein 1979). Therefore, the texture and complexity of channel substrates change in response to altered sediment budgets and flow regimes in developed areas (Paul & Meyer 2001). Despite efforts to control sediment from construction and other ground disturbances, sedimentation is frequently cited as a cause of river impairment in the US (USEPA 2002b, 2006c; Stoddard *et al.* 2005; Paulsen *et al.* 2008). Channel substrates may remain in this condition as rivers respond to increased sediment loads.

In the absence of fine sediments, bed coarsening can occur. As construction continues and new impervious surfaces are created, upslope sediments stabilize, and high streamflows and bank-full flows increase which flush fine sediments from streams and leave larger materials in the streambed (Konrad *et al.* 2005). This process, when localized, is referred to as scouring and can occur over short periods, generally in response to stormflows (Simon & Rinaldi 2006). The long-

term effects of repeated scouring, however, may not be fully evident for decades (Finkenbine *et al.* 2000; Chin 2006). In King County (Washington) the annual number of streamflow events likely to result in scouring was 5- and 11-fold higher in watersheds with 18% and 37% TIA (respectively) compared to forested watersheds (Bledsoe & Watson 2001a). While not reported in detail, May *et al.* (1997) found lowland stream channels that were lacking large wood and had gradients of more than 2% to be more susceptible to stream scour than their undeveloped counterparts. Finkenbine *et al.* (2000) observed reduced amounts of fine substrate material in southwest British Columbia (Canada) streams recovering from the construction phase of urbanization. In a more arid climate, intense summer stormwater discharges flushed fine sediments from the Provo River (Utah) and coarsened stream substrates (Gray 2004a).

Key Findings: Aquatic physical habitat

- Reduced channel and physical habitat complexity are frequent consequences of development. Physical habitat alterations common to urban streams include enlarged channels, eroding banks, excess fine sediment over otherwise coarse streambeds, reduced riparian vegetation and large wood, all of which diminish physical aquatic habitat complexity.
- Construction activities can increase erosion and watershed sediment yields. Changes in sediment budgets and flow regimes associated with development can cause significant changes in channel morphology.
- Accelerated channel erosion can become a significant sediment source for downstream reaches and can propagate the effects of development beyond the reach scale.
- Activities that restructure channels include the installation of pilings, wharfs, and bridges, channel dredging, large wood removal, aggregate removal, bank armoring, and diking and filling in estuaries. Restructuring channels in rivers and estuaries can increase stream power, decrease channel complexity, and increase within-channel erosion.
- Riparian deforestation is often identified as an important factor influencing how aquatic ecosystems respond to development.
- Several mechanisms associated with development reduce potential wood recruitment into streams and the loss of large wood from river channels and estuaries decreases habitat complexity and intensifies the effects of channelization.
- The direction and magnitude of changes to physical habitat depend on the type of stream channel affected, the age of the development, and the spatial pattern and sequence of landscape alterations.

Section 6.0: Fish Passage through Urban and Rural-residential Areas

Movement is a key feature of all salmonid life histories and salmonids exhibit a variety of movements across different spatial and temporal scales. Distances traveled by salmonids range from occasional forays among stream reaches to extensive marine and riverine migrations that traverse entire watersheds and river basins (Quinn 2005). There is an extensive literature documenting the range in salmonid movement distances (Gowan *et al.* 1994; Fausch & Young 1995; Baxter 2002; Rodriguez 2002). Large and small-scale salmonid movements are mediated by differences in body size, physical and physiological requirements of different life stages, different habitat requirements of each life stage, and spatial heterogeneity and connectivity among required habitats (Schlosser 1995). The timing and timeframe over which salmonids make characteristic movements is a product of species-specific life histories and individual differences evolved to maximize survivorship and reproductive success in variable environments.

Movement allows salmonids access to habitats that meet their physiological needs and their requirements for spawning, rearing, and refuge (Schlosser 1995; Pess *et al.* 2002; Rosenfeld *et al.* 2002; Neville *et al.* 2006a; Isaak *et al.* 2007). Travel among habitats is required throughout the lifespans of individual fish. Depending on the species, variable distances separate these habitats. The spatial proximity of distinct habitats and ease of travel among them affects survivorship by controlling the energy required to move between habitats and the predation risk incurred during travel (Dunning *et al.* 1992; Schlosser 1995). Similarly, uninhibited access to functionally similar habitats (e.g., rearing habitat) in different locations allows a population, particularly juveniles, to feed and seek refuge over a broader area and may reduce density-dependent mortality rates (Schlosser 1995). In sum, effective fish passage allows:

- Migrating adults full access to high-quality spawning habitat, resulting in broadly distributed progeny;
- Juvenile access to stream, off-channel, and estuary habitats for rearing;
- Uninhibited downstream movement of smolts as they migrate to the ocean; and
- Fish access to upstream and downstream refuges as chemical, physical, and biological habitats fluctuate.

Fish passage barriers can disrupt landscape-scale habitat connectivity in several ways with varying consequences for population persistence and resilience (Dunning *et al.* 1992; Schlosser 1995; Dunham *et al.* 2003). From a genetic perspective, barriers to movement can disrupt critical dispersal pathways thus isolating populations from one another. Resulting gene flow restriction subsequently erodes genetic variability within populations because incoming migrants no longer offset the processes of genetic drift and inbreeding (Slatkin 1985; Neraas & Spruell 2001; Wofford *et al.* 2005; Neville *et al.* 2006b). If the affected species exhibits metapopulation dynamics (Hanski & Gilpin 1991; Rieman & Dunham 2000), fish passage barriers can also result in local extinctions by altering various demographic and genetic processes (Hanski 1996, 1999). Metapopulations are composed of independent but interconnected populations (of the same species) that occupy a variable number of spatially separate habitat patches at any time (Hanski & Gilpin 1991; Rieman & Dunham 2000). Metapopulations are dynamic and involve movement among habitat patches such that empty patches can be recolonized after populations have gone

extinct. A metapopulation persists when a balance exists between the extinction and recolonization of the constituent populations within the patch network (Harrison 1991). To achieve such a balance, individual fish must be able to move through corridors between patches within the geographical area occupied by the metapopulation. Fish passage barriers that block dispersal corridors sever interconnections among habitat patches, eliminating the opportunity for recolonization following extinction events (Hanski 1996; Rieman & Dunham 2000; Dunham *et al.* 2003).

For salmonids that undertake long-distance marine or riverine migrations, habitat connectivity is synonymous with fish passage across entire watersheds or river basins. Rivers must be barrier free for their entire lengths between estuaries or river mainstems and spawning areas (adult upstream migration), and between rearing areas and estuaries or river mainstems (juvenile or adult downstream migration). Thus, both resident and migratory salmonids require the freedom to move among stream reaches that contain habitat suitable for different behaviors (feeding vs. spawning), high and low flow refuges, and life stages (juvenile vs. adult). For all salmonids, life history variation results in species-specific differences in barrier passage ability (Myers *et al.* 2006) and varying consequences for landscape-level fragmentation caused by fish passage barriers (Sheer & Steel 2006). Barrier-free waters are also critical for many non-salmonid fish that migrate long distances in rivers, such as lampreys, sturgeons, suckers, and minnows.

Section 6.1: Factors and Mechanisms that Alter Fish Passage in Developed Areas

Across the Pacific Northwest, instream barriers (e.g., culverts, dams) are known to block anadromous fish access to entire basins and have resulted in basin-wide extirpations (Lichatowich 1999; Beechie *et al.* 2006; Sheer & Steel 2006). These barriers also fragment salmonid populations, limiting gene flow, reducing genetic diversity (Neraas & Spruell 2001; Wofford *et al.* 2005), potentially reducing population persistence through landscape scale disturbances (Schlosser 1995), and ultimately increasing extinction risk. Urban and rural-residential developments along streams, rivers and estuaries in Oregon contain an assortment of fish passage barriers that may prevent salmonids from reaching habitats for feeding, rearing, acclimating, and spawning. This section describes the degree to which rural-residential and urban developments prevent or hinder salmonids from reaching critical habitats.

Decreased fish passage in developed landscapes results from several human mediated actions that create both physical and physiological barriers including:

- Inadequate or excessive streamflow and velocities,
- Culverts and other in-stream structures,
- Dams,
- Streams piped underground,
- Artificial lighting,
- Noise, and
- Poor water quality.

Streamflows – Sufficient streamflow is an obvious requirement for salmonid movement. Reduced flows can block or delay movement (Myers *et al.* 2006) and increased peak flows can increase mortality by sweeping eggs or fish downstream during critical periods of salmonid life cycles. As discussed in Section 4.0 of this report, both of these hydrologic conditions result from rural-residential and urban development. In some cases, flows may be adequate for fish passage through pre-disturbance stream channels, but water in channels reshaped by development may flow at depths insufficient or velocities excessive for fish migrations.

Culverts – Culverts and similar devices create a pervasive salmonid habitat connectivity problem throughout all land uses in Oregon (GAO 2001; Heller & Sanchez 2005; Sheer & Steel 2006). Several aspects of culvert design result in obstructed fish passage (Bates *et al.* 2003; Clarkin *et al.* 2005). Examples include high water discharge points or shallow plunge pools, high water velocity and turbulence within the culvert, inadequate depth within the culvert, obstructions within or at the upstream end of the culvert, or water flow under or around the culvert. In addition to the high density of road crossings, the hydrologic regime changes that accompany development can intensify channel degradation around culverts, increasing the likelihood that these structures will become fish passage barriers (Bates *et al.* 2003).

Dams – Hydropower dams create well documented fish passage problems (e.g., Ruckelshaus *et al.* 2002 and references therein). In the Pacific Northwest, hydropower projects service the energy demands of a growing population and their environmental effects are part of the ecological footprint of development. More directly linked to urban and rural-residential areas however, are smaller dams that impound public water supplies. Few of these dams incorporate effective fish passage structures. As a result, fish are blocked from accessing miles of habitat both upstream and downstream of these dams. Ironically, habitat upstream of these barriers typically is high quality because of efforts to ensure that public water supply watersheds are relatively undisturbed.

Streams Piped Underground – Probably the most thorough aquatic alteration practiced during the development of older urban areas was piping streams underground (e.g., Metro 1999). This involved diverting long segments of streams from their natural channels into pipes to route the water to larger waterways. In these piped streams, inadequate streamflow (depth, velocity) and other obstructions (e.g., length, lack of light) severely limit or prevent fish passage. Ecological functions in the piped reaches have also been largely eliminated and several authors have suggested that extensive underground piping of headwater streams can alter hydrology, water quality and nutrient cycling in downstream river reaches (e.g., Alexander *et al.* 2007; Freeman *et al.* 2007; Meyer *et al.* 2007; Wipfli *et al.* 2007). The net result is lost access to upstream habitats required by migratory fish, lost habitat in the portion of the stream system that is underground, and alterations to habitat and aquatic species assemblages in downstream reaches.

Artificial Lighting – In laboratory, artificial stream, and field experiments (Cedar River, Washington), Tabor *et al.* (2004) reported that intense light at night may inhibit sockeye salmon fry migration and may increase predation by freshwater sculpins (*Cottus* spp.) as the fry slow or stop their outmigration until light conditions are more favorable. The overall effect, most likely localized, on sockeye salmon fry could not be fully assessed. It is not known how intense artificial lighting associated with development and transportation corridors may affect other salmonid species.

Noise – There is some concern that excess noise generated by anthropogenic sources (e.g., active bridge and pier construction, pile driving, traffic noise) might affect the health or survival of fishes by causing hearing loss or altering ecologically important behaviors (Song *et al.* 2008). The few studies conducted on this phenomenon show results ranging from damaged auditory structures and prolonged hearing loss to unaffected auditory structures and temporary hearing loss. These differences may be specific to individual fish species or to the water depth or noise amplitude used in different experiments (reviewed by Song *et al.* 2008). However, no research has addressed whether noise in the auditory range of fish could constitute a temporary or permanent barrier to fish migration or might otherwise alter the migratory behavior of either juvenile or adult fish.

Poor Water Quality – Water quality is an important attribute of dispersal and migration routes. High turbidity, excessive temperatures, low levels of dissolved oxygen, and pollutants effectively block movement by presenting physiological and olfactory challenges to fish (Myers *et al.* 2006). Water quality is discussed in Section 7.0 of this report, and will not be discussed further in this section.

Section 6.2: Nature and Magnitude of Fish Passage Changes in Oregon

The timing and magnitude in which passage barriers block fish movements among critical habitats vary in biologically important ways. Complete barriers block all fish movements throughout the year regardless of flow velocity and volume. Temporal barriers block all fish movements some of the time and can delay upstream or downstream movements of adults or juveniles depending on streamflow. Partial barriers block only some fish (e.g., smaller individuals or species) some or all of the time and can be among the most difficult to identify as barriers during inventories and assessments such as those conducted on culverts (Clarkin *et al.* 2005).

The degree to which Oregon's salmonid populations are limited by fish passage barriers associated with rural-residential and urban areas is poorly documented. Complete barriers formed by some dams are obvious. For example, the City of Portland constructed the Headworks Dam in 1922 to increase the municipal water supply from the Bull Run River. The dam is known to block approximately 37 miles of salmon and steelhead habitat (Taylor 1998). The magnitude of cumulative passage impacts from temporal and partial barriers of broadly distributed culverts, underground piping, and water quality are more difficult to estimate. Within the Willamette and lower Columbia River basins, Sheer & Steel (2006) identified 1,491 complete fish passage barriers blocking access to 9,277 stream miles, approximately 42% of historic stream habitat. These barriers prevent anadromous fish from reaching 40% of streams with gradients suitable for steelhead, 60% of streams with habitat in good condition, and 30% of streams draining watersheds dominated by coniferous land cover. Not surprisingly, the largest proportional lengths of blocked habitat occur in watersheds where large hydroelectric dams are present (Sheer & Steel 2006).

Many fish passage barriers are clustered near roads in areas that contain most of Oregon's rural-residential and urban developments (i.e., lower elevations and within floodplains). The high road density in rural-residential and urban areas increases the likelihood that culverts and other structures restrict or slow salmonid movements through developments. Between 1996 and 1999,

ODFW and ODOT conducted a collaborative survey of state and county owned roads in Oregon and evaluated over 5,500 culverts for their potential to pose barriers to fish passage (Mirati 1999). The surveys did not include roads owned by private (e.g., forestlands, residential property), federal³³, or city entities (Mirati 1999). Fish were not able to pass through approximately 52% (~2,870) of the inventoried culverts. Oregon's coastal and upper Willamette Valley regions have the majority of examined culverts (38% and 33% respectively) and identified barriers (50% and 27% respectively)³⁴. The IMST was not able to determine the degree to which these roads and fish passage barriers were associated with rural-residential and urban developments in Oregon.

Metro³⁵ (Oregon) conducted a culvert survey in 1999 and 2000 in an area roughly within the urban growth boundary and found 1,500 culverts that had not been documented in other survey efforts, including the collaborative effort by ODFW and ODOT. One hundred fifty (150) or 10% of these act as complete barriers to fish movement (Metro 2002). The IMST was not able to locate similar data for other urban areas in Oregon.

Within the city of Portland, 41 miles of streams (18% of the 227 stream miles) are known to be piped underground (City of Portland 2004). These alterations are most numerous in tributaries on the eastern side of the Willamette River, but also include streams on the western side (Metro 1999). The entire lower Willamette River watershed is estimated to contain 182 miles of piped streams but the proportion of total stream miles is unknown because historical stream location and piping records are incomplete (City of Portland 2004). The IMST was not able to locate data on piped streams or otherwise lost habitat in other developed areas in Oregon, but it is logical to assume that the problem is similar in areas that were established and urbanized over timeframes similar to Portland (e.g., Salem, Eugene, Roseburg, Medford, Coos Bay).

Together, culverts, piped streams, and similar development practices have eliminated approximately 400 stream miles from the Portland-Metro region (Metro 1999). The degree to which these streams supported salmonids is unknown. Given the wide distribution of salmonids in other streams in the Portland-Metro region, however, it is likely these eliminated streams did support salmonids, both directly by providing habitat and indirectly as conduits for water and energy to downstream locations used by salmonids.

³³ The US Bureau of Land Management and US Forest Service estimated that as of 2001, as many as 5,500 road-associated culverts blocked fish passage on public lands in Oregon and Washington.

³⁴ Data available at <http://www.dfw.state.or.us/fish/passages/inventories.asp>. Accessed on December 26, 2010.

³⁵ Metro (<http://www.oregonmetro.gov>) is the elected government responsible for managing a regional urban growth boundary encompassing 25 cities and more than 60 special service districts within Washington, Multnomah and Clackamas counties (Oregon).

Key Findings: Fish passage

- Urban and rural-residential developments along streams, rivers and estuaries in Oregon contain an assortment of fish passage barriers that may prevent salmonids from reaching habitats for feeding, rearing, acclimating, and spawning.
- Decreased fish passage in developed landscapes may result from inadequate or excessive streamflow and velocities, culverts and other in-stream structures, dams, piped streams, artificial light, excessive noise, or poor water quality.
- The degree to which Oregon's salmonid populations are limited by fish passage barriers associated with rural-residential and urban areas is poorly documented. Complete barriers are formed by some dams used to store municipal water supplies. However, the magnitude of cumulative passage impacts resulting from broadly distributed culverts, piped streams, and water quality barriers are more difficult to estimate.
- Estimates from the Portland Metro region indicate that fish barriers formed by culverts and streams piped underground may be extensive throughout Oregon's urban areas, but similar information for these other areas were not located.

Section 7.0: Water Quality

Since the US Congress passed the Clean Water Act (CWA) in 1972, significant progress has been made in identifying and reducing point source pollution discharges into US surface waters (Beasley & Kneale 2002; Mrazik 2006). However, the USEPA recently assessed approximately one fifth of the nation's waters and found that 39% do not meet state-level water quality standards (USEPA 2002b; GAO 2004). In a survey of the coastlines and estuaries of the conterminous US, USEPA (2004a) reported that 60% of the near coastal surface area failed to attain reference conditions³⁶ for water quality and 24% failed for sediment chemistry. Following a survey of all wadeable streams in the conterminous US, the USEPA (2006c; Paulsen *et al.* 2008) concluded that 67% of the total stream length failed to attain reference conditions. Among the limited number of water quality variables measured, streams with excess nitrogen, phosphorus, and fine sediments were most often associated with biological assemblages in poor condition.

Many waters remain impaired by non-point pollution sources that are difficult to identify and control (NRC 1992; USEPA 1996; Carpenter *et al.* 1998; Wentz *et al.* 1998; Beasley & Kneale 2002; ODEQ 2004; Brett *et al.* 2005a, b; Carle *et al.* 2005; Atasoy *et al.* 2006; Mrazik 2006). Between 1991 and 2001, the USGS National Water-Quality Assessment Program (NAWQA; Hamilton *et al.* 2004) characterized water quality in 51 major river basins draining approximately half of the US land area. The following major findings were most relevant to rural-residential and urban developments.

- Surface and groundwater contamination is widespread in urban areas.

³⁶ Reference conditions reflect the range of natural variability characterized from or thought to represent an ecosystem under least-disturbed conditions.

- Without exception streams and groundwater in areas with significant urban development contained a complex mixture of nutrients, trace elements, pesticides, volatile organic compounds, and the chemical breakdown products of these constituents.
- Breakdown products were found as frequently as their parent contaminants.
- New products that have the potential to become water contaminants in urban environments are approved for use each year.
- Many contaminant concentrations varied seasonally, with long periods of low or undetectable levels punctuated by short periods of high concentration.
- Organochlorine compounds no longer in use (e.g., DDT, PCBs³⁷) were detected in 95% of fish tissue samples collected at urban sites and exceeded tissue concentrations determined to be protective of wildlife at 75% of urban sites.

In Oregon, 13,937 stream miles monitored by ODEQ fail to meet one or more water quality standard while only 5,687 miles met all standards (Oregon Progress Board 2000)³⁸. Where urban lands dominated the 5-mile radius surrounding a stream monitoring site, water quality rankings fall in the poor or very poor range (Figure 7-1; Oregon Progress Board 2000). Data from four Oregon river basins indicated that while many pollutants are discharged primarily from point sources, non-point sources contribute a majority of the pollutants that exceed Oregon's water quality standards (Oregon Water Progress Board 2000). While the non-point contributions of individual urban areas vary, they can pose a significant threat to water quality (NRC 1992; USEPA 1996; Oregon Progress Board 2000; Brett *et al.* 2005a, b; Atasoy *et al.* 2006).

³⁷ Polychlorinated biphenyls are a class of organic compounds that includes over 200 closely related chemical compounds with variable toxicities.

³⁸ Oregon contains approximately 114,500 linear miles of rivers and streams, only a small fraction of which are monitored by ODEQ (Oregon Progress Board 2000).

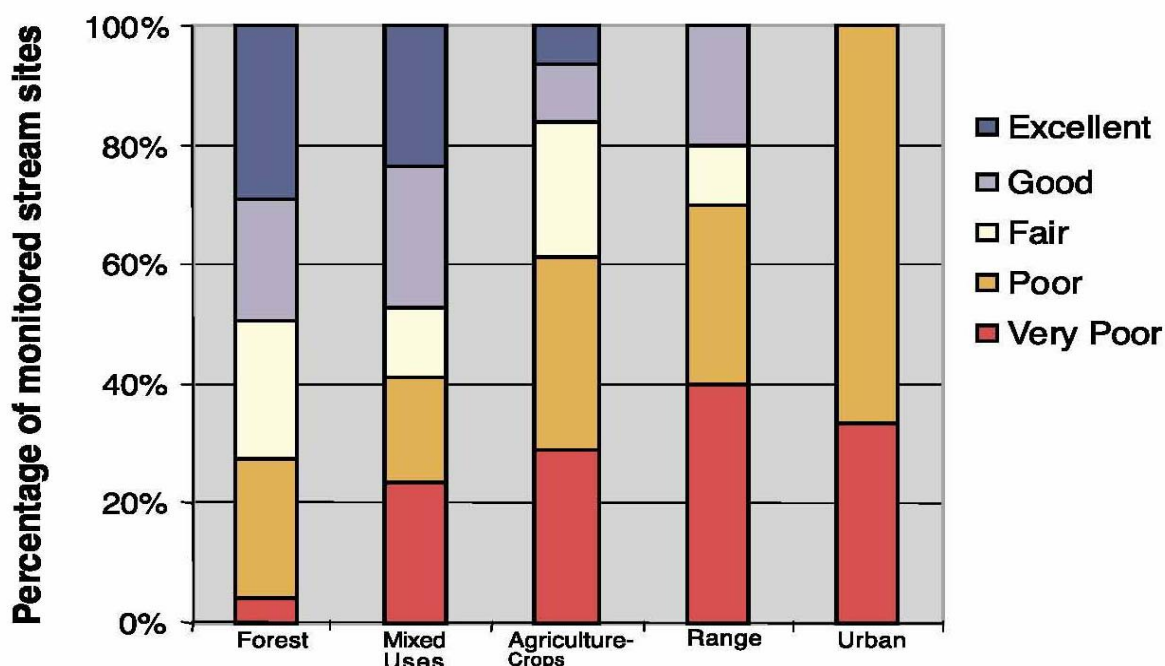


Figure 7-1. The relationship between water quality and adjacent land use in Oregon. Water quality, measured using the Oregon Water Quality Index, integrates information on temperature, dissolved oxygen, biochemical oxygen demand, pH, nutrients, total solids, and fecal coliform bacteria. Land use categories are based on dominant land use within a five-mile radius of the monitoring site. Streams in Oregon's urban areas typically have water quality problems related to temperature and dissolved oxygen. Additionally, some streams are adversely affected by oxygen demanding wastes (measured as biological oxygen demand) and by excess nutrients, usually phosphorous and nitrogen. Figure reproduced from the Oregon State of the Environment Report (Oregon Progress Board 2000).

Section 7.1: Sources of Variation in Water Quality Response to Development

Several mechanisms operating in rural-residential and urban developments increase the likelihood of water quality impairment (Figure 7-2). The diversity and quantity of nutrients and compounds (e.g., fertilizers, pet waste, wastewater effluent, increased erosion) available for transfer to streams increase dramatically with development. In rapidly growing areas, runoff from construction sites can be a significant sediment source despite control measures. Chemicals toxic to fish and other aquatic biota originate from a variety of point and non-point sources. The density of point sources and the volume of effluent they produce increase as urban populations grow. Removal and disturbance of riparian vegetation reduce the capacity of riparian areas to sequester pollutants and sediments before they reach streams. Stormwater runoff across impervious surfaces transfers pollutants directly to streams and estuaries, bypassing soils and remaining vegetation that could otherwise filter, sequester, and break down potential water contaminants.

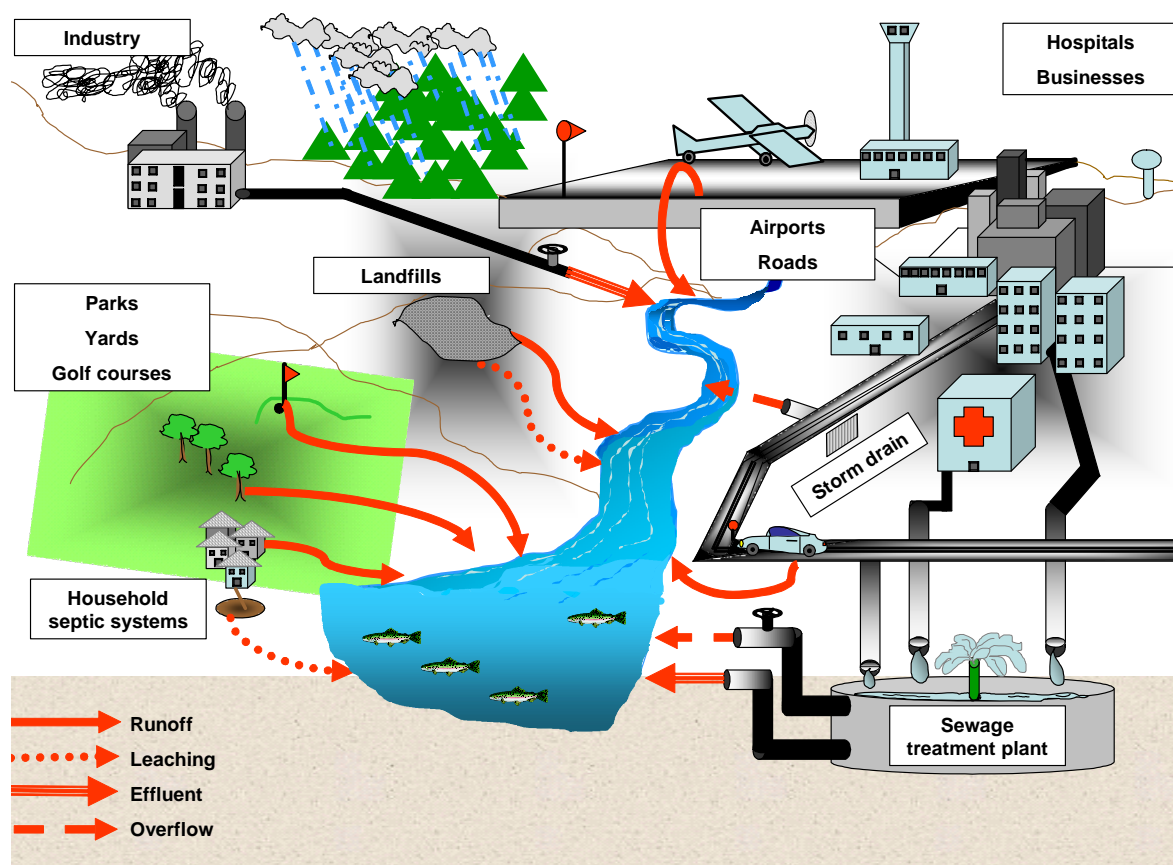


Figure 7-2. A generalized schematic showing sources of pollutants in the urban landscape. Note that the figure is not drawn to scale and that the potential pollutant contributions by each category are not drawn proportionately to the relative size of impact.

Quantifying the association between specific elements of water quality and rural-residential or urban land uses is difficult because water quality parameters are affected by various natural and anthropogenic factors (Coulter *et al.* 2004). Natural features (e.g., topography, geology, climate, hydrology, soils) underlying developments control concentrations of naturally occurring substances that may be considered contaminants when they exceed ‘background’ levels (Hamilton *et al.* 2004). Natural features also affect how contaminants are transported from land to water and create variability in how contaminants accumulate in different environments. Stream discharge can play a major role in the magnitude of pollutant concentration in surface waters (Lehrter 2006). Small streams respond rapidly to storm events resulting in contamination peaks that rise and fall rapidly compared to larger rivers, which maintain lower contaminant concentrations for longer periods of time (Hamilton *et al.* 2004). Surface water is typically more vulnerable to contamination than groundwater which is somewhat protected by overlying substrates that filter contaminants and where relatively long residence times may allow some chemicals to degrade or disperse. Consequently, it may take decades for groundwater to show impairments (Hamilton *et al.* 2004); once contaminated, however, groundwater is extremely difficult to decontaminate. Groundwater may be more vulnerable in areas with permeable soils or

areas where groundwater pumping lowers water tables, induces surface water infiltration, and thus transports surface contaminants to groundwater (Hamilton *et al.* 2004). This amount of natural variability can cause water quality to fluctuate within single rainfall events, seasonally, annually, or over variable intervals, depending on location (geographic, surface, or subsurface). Connecting changes in water quality to specific land uses requires that long-term trends be distinguished from natural fluctuations. The historical data required for such analyses are lacking in many areas of the Pacific Northwest (Hamilton *et al.* 2004).

The intensity, type, and distribution of development also introduce variability in the nature and magnitude of water quality degradation. Schiff & Benoit (2007) found that their integrative water quality index was most strongly associated with impervious surface area, highlighting the role impervious surfaces play in delivering non-point pollution to streams. Similarly, Atasoy *et al.* (2006) found that a 1% increase in effluent discharges from point sources had a much smaller impact on total phosphorus, nitrogen, and suspended solids than a comparable increase in land uses that generate non-point source pollution. However, the percentage of impervious surface cover associated with measurable water quality impairment ranges from 4–5% (May *et al.* 1997) to 10–12 % (Klein 1979; Wang *et al.* 2000). Much of the variation observed in the relationship between impervious surfaces and water quality depends on the quantity of non-point contaminants washed from impervious surfaces, connectivity between impervious surfaces and streams, how well riparian buffers shield streams from non-point source pollution (Ourso & Frenzel 2003; Hatt *et al.* 2004), and on previously existing land uses such as agriculture and mines (ODEQ 2006; see Section 1.41 of this report).

The stage of development also introduces variability in the water quality response. The types and quantities of contaminants that enter streams when undeveloped lands are converted to residential uses (i.e., new construction) can be qualitatively different from contaminants derived from subsequent uses of converted land. Former agricultural lands may have pesticides and associated metals in the soil, groundwater, and sediments (e.g. in drainage ditches) at concentrations above acceptable risk levels (as defined in Oregon Revised Statute 465.315; ODEQ 2006). Atasoy *et al.* (2006) found that both land conversion and developed land use increased total phosphorus and total nitrogen loadings in North Carolina streams. In their study, only land conversion (i.e., new construction) increased total suspended solids (despite unspecified standard control practices). New construction also had a larger effect on total phosphorus than existing developments (Atasoy *et al.* 2006), possibly because phosphorus binds to fine sediments. Carle *et al.* (2005) found that water quality variation was best explained by development type, connectivity to city stormwater and wastewater systems, and development density, indicating that older, densely developed neighborhoods and younger, low-density suburban developments contribute to water quality impairment in different ways.

The pattern in which different pollutants accumulate and move into streams is also variable and dependent on seasonality of rainfall. Many pollutants (e.g., suspended solids, various nutrients, metals) follow a ‘first flush’ pattern where runoff generated early in a rainfall event transports most of the total pollutant load and is the most contaminated (Lee *et al.* 2002). Quantitative definitions of this phenomenon generally identify a fraction of the total pollutant load (e.g., >50%) that occurs in an initial fraction (e.g., 25%) of storm event runoff (Flint & Davis 2007). In climates like that of the Pacific Northwest where rainfall occurs during distinct seasons, the first flush concept can also apply to the rainfall season (i.e., the earliest storms transport the majority of the available pollutant loads). During a first flush event, a disproportionately high quantity of

pollutants discharge into surface waters but individual pollutants exhibit variable peak loads within and among storm events (Chang & Carlson 2005; Flint & Davis 2007). In an analysis of 38 storm events in 13 separate urban watersheds, Lee *et al.* (2002) found that the contaminants dominating first flush events depended on both land use (i.e., residential vs. industrial) and analysis method and that first flush intensity increased as watershed size decreased and rainfall intensity increased. Within a given storm event, some pollutants may also undergo a 'second flush,' defined as flushing of 50% of the total pollutant load in any 25% portion of the runoff volume beyond the first 25% (Lawler *et al.* 2006; Flint & Davis 2007). Monitoring designs for municipal stormwaters are unlikely to detect the highest pollution loads if sampling designs do not account for first and second flush phenomena.

The following sections summarize the effects of development on several water quality parameters including suspended sediment and turbidity, nutrients, water temperature, dissolved oxygen, and toxic pollutants. Concluding remarks on the responses of aquatic biota, including salmonids to changes in water quality and the potential for rehabilitation of water quality follow these sections.

Section 7.2: Suspended Sediment and Turbidity in Urban and Rural-residential Areas

In this section, IMST describes the factors and mechanisms that alter suspended sediment and turbidity as well as the nature, timing, and magnitude of suspended sediment and turbidity changes in surface waters affected by development.

SECTION 7.21: FACTORS AND MECHANISMS THAT ALTER SUSPENDED SEDIMENT AND TURBIDITY

Sediments transported by streams and rivers serve integral roles in many ecosystem processes to which salmonids are adapted. Periodic flooding, associated with increased erosion of sediments, delivers and sorts gravel required for salmonid spawning. Sediment loads commonly reflect the location, quantity, and composition of sediment sources found throughout the contributing watershed. Geology, vegetation cover, landscape topography, and weather conditions determine erosion patterns of sediment, ranging from fine (e.g., clay, silt) to coarse (e.g., gravel, cobble) from upland slopes and stream channels. Once sediments enter a stream, many factors influence whether they remain suspended within the water column or are deposited, including particle size, channel gradient, water velocity, and the total sediment load carried by the stream. Fine sediments that remain suspended can be particularly damaging to aquatic organisms and even low-level accumulation of fine sediments in stream beds can reduce salmonid spawning success and alter fish and macroinvertebrate assemblages (Bryce *et al.* 2008, 2010). Water-borne clay and silt influence numerous aspects of water quality when they bind to phosphorus and toxic contaminants (e.g., heavy metals) and facilitate transport of these substances to streams (Brett *et al.* 2005a; Lawler *et al.* 2006; Li *et al.* 2006).

Turbidity, a measure of light transmission through water, is an important water quality variable (Lawler *et al.* 2006). Increasing turbidity reduces light penetration through water and, consequently, photosynthetic productivity that forms the basis of many aquatic food webs (reviewed by Henley *et al.* 2000). Through its relation to light suppression, turbidity can

influence the biological oxygen demand and dissolved oxygen available in streams (see Section 7.5). Many factors influence turbidity including suspended sediments, water chemistry, terrestrial plant debris, and aquatic microorganisms (Lloyd 1987; Henley *et al.* 2000). Water containing excess fine sediment is highly turbid. In combination, increased suspended sediment and turbidity can lead to population and community-level changes in aquatic ecosystems (Lawler *et al.* 2006).

Urban and rural-residential development increases both suspended sediment and turbidity through multiple pathways with numerous detrimental effects on salmonids and other aquatic biota (Paul & Meyer 2001; Opperman *et al.* 2005; Lohse *et al.* 2008). Direct influences are most obvious when land-clearing and construction activities mechanically disturb large quantities of soil (Wolman & Schick 1967; Keller 1999; Paul & Meyer 2001; Atasoy *et al.* 2006). Manipulation of the stream channel during development also increases suspended sediment concentrations. Removal of riparian vegetation and large wood, channelization, bridge construction, and dredging can cause significant stream channel erosion (Booth 1991; Waters 1995; Keller 1999). Road networks that expand along with rural-residential and subsequent urban development are a chronic source of fine sediment delivery to streams (Carter *et al.* 2003). Indirect influences are a function of hydrologic and physical habitat changes that accompany development (see Sections 2.31 and 2.32 of this report). For example, channelization and bank stabilization confine rivers and streams to a single channel, diminishing sediment filtration and storage by riparian zones (see Section 2.32 of this report). None of these factors operate in isolation and instead act in a cumulative manner with other land uses to increase sediment erosion and, consequently, to increase suspended sediment and turbidity.

SECTION 7.22: NATURE, TIMING, AND MAGNITUDE OF SUSPENDED SEDIMENT AND TURBIDITY CHANGES

Sediment eroded from the landscape is one of the most ubiquitous non-point source pollutants affecting surface water quality (Nelson & Booth 2002). Few studies have estimated the magnitude of sediment generated by development in Oregon. In Fanno Creek (primarily located in southwestern Portland, Oregon), runoff from urban and industrial lands can contribute as much as 2.5 million pounds of sediment annually (ODEQ 2001b). A 1997 study commissioned by Oregon's Governor found that, on a per acre basis, urban sites in the Willamette River basin contributed the greatest fraction of suspended sediment to the river (GAO 1998). Research conducted elsewhere demonstrates that total suspended solid (TSS) concentrations and turbidity vary considerably with development. In rapidly developing Issaquah Creek (western Washington), Nelson & Booth (2002) estimated that channel erosion associated with urban development increased watershed-wide sediment production by 20%, with an additional 12% increase contributed by other urban impacts including construction, road surface erosion and sediment production from residential and commercial areas.

In contrast, Brett *et al.* (2005a) found little association between urban land cover and either TSS or turbidity in 17 Seattle (Washington) area watersheds. Burcher & Benfield (2006) found that TSS concentrations were significantly lower in North Carolina suburban developments compared to other land uses. A common explanation for such observations is that watersheds contribute increased sediment loads to streams during the construction phase of development, but TSS concentrations decrease relative to predisturbance conditions after impervious surfaces cover

potential sediment sources (Finkenbine *et al.* 2000; Atasoy *et al.* 2006). For example, Carle *et al.* (2005) found that low TSS concentrations in North Carolina streams were associated with impervious surface area connections to stormwater sewer systems (as measured by effective impervious area or EIA) in older sections of the city of Durham where new construction was limited. In new developments, high TSS concentrations were attributed to erosion from construction sites, despite application of stormwater regulations (Carle *et al.* 2005).

Discrepancies among studies characterizing the relationship between development and TSS concentrations or turbidity may result from differences in study approach or methodology, type of disturbance, disturbance duration, disturbance proximity to stream, watershed size, vegetation characteristics, climate, or geography (Nelson & Booth 2002; Coulter *et al.* 2004). In mixed-use watersheds (e.g., urban, agricultural, undeveloped) the percentage of different land use types is important because they differ in their non-point source pollution contributions, particularly suspended sediments (Coulter *et al.* 2004). Given the discrepancies among available research results, IMST, in the text below, characterizes the nature, magnitude, and timing of sediment and turbidity changes in relation to 1) the development phase, 2) a single storm event, and 3) timing in the annual cycle.

During Development – Development initiates or modifies a number of sediment producing processes including construction site erosion, road surface erosion, and channel-bank erosion (Nelson & Booth 2002). As discussed in Section 2.0, new construction triggers substantial changes in sediment supply and annually produces ~ 80 million tons of sediment that enters US waters (Harbor 1999). Results from 100 published studies, conducted worldwide over the past 50 years, show that sediment production in urbanizing watersheds can increase from 2- to 10-fold during the initial phases of development (Chin 2006).

While there is abundant information on the amount of sediments entering aquatic ecosystems, only a few studies have documented a direct linkage between residential or commercial construction and changes in measured TSS or turbidity in surface waters. Monitoring conducted during 1997 by the City of Portland (2006) showed that construction sites release large amounts of sediment. One monitored site released more than three times the amount of TSS in a single storm event compared with that released by a reference site over the course of a year, which helped trigger Portland to develop and implement new city codes and develop an erosion control manual to better control erosion during construction. Olding *et al.* (2004) found that during moderate to large rainfall events, stormwater management facilities operating in sites with new construction had TSS discharge 2 to 3 times higher than identical facilities operating in more stable watersheds.

Many studies have documented how altered sediment budgets and hydrology typically contribute to channel erosion (Paul & Meyer 2001; Nelson & Booth 2002; Chin 2006) which then becomes a source of increased TSS concentrations and turbidity in downstream reaches. This is particularly true in valley-bottom channels that are easily eroded and susceptible to channel enlargement from increased discharge (Nelson & Booth 2002). Development may result in decreased surface erosion rates once impervious surfaces cover large areas, but hydrologic changes stemming from development also increase sediment loads delivered downstream by increasing channel erosion. For example, in a southern California watershed with 50% urban

cover, Trimble (1997) estimated that 67% of total sediment production came from channel-bank erosion within San Diego Creek.

During a Storm Event – Schiff & Benoit (2007) reported that turbidity and TSS concentrations fluctuate by orders of magnitude during individual storm events. Lawler *et al.* (2006) characterized turbidity dynamics in a United Kingdom headwater stream (Tame River, a tributary of the Thames River) after rainfall events and found that TSS concentrations increased as individual storms progressed and throughout storm events sequences. Similarly, Flint & Davis (2007) found that suspended solids frequently exhibited a second flush that contained higher sediment loads than the first flush from the same storm. Lawler *et al.* (2006) attributed increasing turbidity within a storm event to suspended solids from either combined sewer overflows or more distant sediment sources arriving at monitoring sites during latter stages of the storm. These authors interpreted higher turbidities during successive storms as evidence that sustained supplies of suspended solids can exist in some urbanized systems. In an arid ecosystem (Provo River, Utah), Gray (2004a) observed that larger summer storms generated TSS flushing events with poor water quality conditions persisting up to 12 hours. Storms of smaller magnitude did not alter water quality parameters beyond the natural range of variation for the stream (Gray 2004a). Sansalone & Cristina (2004) and Cristina & Sansalone (2003) described storm event flushing of TSS and total dissolved solids as flow limited. In other words, higher flow events (per unit drainage area) exhibited a ‘front-loaded’ first flush while low flow events flushed particulate matter more consistently over time.

Throughout the Annual Cycle – The annual timing of sediment movement into streams is dependent on hydrology, rainfall patterns, and timing of sediment producing activities such as construction. In undisturbed watersheds of western Oregon, the first one or two large rainfall events in the fall are likely to transport much of the sediment that enters streams during any given year (GAO 1998). In contrast, urban runoff can transport sediment to streams at any time of year (ODEQ 2001b). Most of the sediment flushed from urban surfaces consists of fine particles that bind to both urban-generated and natural (or background) pollutants. To prevent these pollutants from reaching surface waters, many municipalities and stormwater districts often detain urban runoff to allow sediments to settle out of the water column. As a result, much research attention has been focused on documenting TSS flushing behavior and efficiency of settling ponds under variable rainfall conditions at different locations (e.g., Comings *et al.* 2000; Bledsoe 2002; Hossain *et al.* 2005; Bäckström *et al.* 2006; Birch *et al.* 2006; Li *et al.* 2006; Kang *et al.* 2007). Of particular interest are sediments washed from road surfaces because they are smaller than those typically trapped by stormwater detention structures, and are more likely to transport nutrients, metals and toxic contaminants to streams (Li *et al.* 2006). In their study, Li *et al.* (2006) found that two-compartment settling tanks, when capturing and retaining the first 20% of runoff volume, could remove 40% of the total particulate load. In another study, Birch *et al.* (2006) found TSS retention by a pond next to a roadway during storms varied from below 0 (i.e. the pond was a source of TSS) to as high as 93%, depending on rainfall conditions. Resuspension and flushing of sediment during storm events can be a confounding factor, reducing the effectiveness of stormwater detention structures (Norton 2008). As many of these studies document both highly variable TSS concentrations entering stormwater detention structures and highly variable TSS removal efficiency (e.g., Bäckström *et al.* 2006; Birch *et al.* 2006; Li *et al.*

2006) it is difficult to draw any general conclusions of expected retention ranges for TSS relevant to developments in Oregon.

Key Findings: Sediments

- Urban and rural-residential development increases both suspended sediment and turbidity through land-clearing and construction activities, manipulation of the stream channel, and removal of riparian vegetation.
- None of the factors affecting sediments operate in isolation. Instead they act in a cumulative manner with other land uses to increase sediment erosion, and consequently, to increase suspended sediment and turbidity.

Section 7.3: Nutrients in Urban and Rural-residential Areas

In this section, IMST describes the factors and mechanisms that alter nutrient concentrations and the nature, timing, and magnitude of nutrient changes in streams, rivers, and estuaries affected by development.

SECTION 7.31: FACTORS AND MECHANISMS THAT ALTER NUTRIENT CONCENTRATIONS

Rural-residential and urban developments function as nutrient sources worldwide (Kaye *et al.* 2006). Because nitrogen (N) and phosphorus (P) frequently become pollutants in aquatic ecosystems affected by development, IMST's review focuses primarily on these nutrients. In ecosystems that are not limited by light, the availability of N and P is a strong determinant of photosynthetic rates, plant growth, and biomass production of organisms that form the foundation of food webs. When developed landscapes supplement nutrient delivery to surface and ground waters, phytoplankton and aquatic plant production (including nuisance and toxic algal blooms) can increase dramatically. This process known as eutrophication is a common cause of surface waters failing to meet CWA water quality standards (USEPA 1996, 2002b; Brett *et al.* 2005a) and is strongly associated with poor biological condition (as reflected in macroinvertebrate assemblages) in streams nationwide (Paulsen *et al.* 2008).

Anthropogenic nutrient sources (particularly landscaping fertilizers, septic fields, detergents in municipal wastewater effluent, and commercial and industrial discharges) increase the total amount of N and P available for transport to streams (Bowen & Valiela 2001; Brett *et al.* 2005a, b; Atasoy *et al.* 2006). The nutrient form (e.g., ammonium vs. nitrate and nitrite for N) and magnitude of increase depend largely on patterns of septic tank discharge, wastewater treatment technologies, fertilizer use, and atmospheric deposition including automobile exhaust (reviewed in Bowen & Valiela 2001; Paul & Meyer 2001; Bernhardt *et al.* 2008). Nitrates are easily leached from the soil and transported to groundwater (p. 167, Novotny 1995). Based on the US National Stormwater Quality Database³⁹, freeways produced the largest concentrations of

³⁹ Database only included land uses in developed areas. Agriculture was not included.

ammonia and total Kjeldahl N in stormwater, industrial sources accounted for the largest concentrations on nitrate and nitrite, whereas residential areas had the largest concentrations of total N (Bernhardt *et al.* 2008)⁴⁰.

Soluble forms of P are also transported to streams as wastewater from septic, municipal, commercial, and industrial sources as well as atmospheric deposition (e.g., Schuster & Grismer 2004; Brett *et al.* 2005b). Phosphorus applied as fertilizer is commonly bound to sediment particles (p. 166, Novotny 1995), therefore processes that accelerate erosion and increase suspended sediment transport to streams also increase particulate P transport (Brett *et al.* 2005b). After entering surface waters, the size of the sediment particle to which the P is attached determines how fast the particulate P settles out of the water column or remains suspended (i.e., organic matter, clays, silts) where the P is biologically available to aquatic plants (Brett *et al.* 2005b; Ellison & Brett 2006). Many early studies linked increased total P to particulate-bound nutrients mobilized during stream bank erosion and movement of in-stream sediments (reviewed by Paul & Meyer 2001). Based on the US National Stormwater Quality Database, industrial sources accounted for the largest concentrations of orthophosphate⁴¹ in stormwater, while residential areas had the largest concentrations of dissolved and total P (Bernhardt *et al.* 2008).

In intact terrestrial ecosystems, riparian and upland vegetation can play a significant role in capturing and sequestering excess nutrients from stormwater runoff or floodwaters. However, extensive manmade drainage and flood control systems effectively bypass the beneficial services of soil microbes and vegetation present in riparian and upland areas (Brett *et al.* 2005a; Cadenasso *et al.* 2008). Impervious surfaces accumulate deposits of various N and P compounds which are then washed directly into waterways rather than entering the soil where they can be removed by plant and microbial processes (Coats *et al.* 2008). Bypassing vegetated areas decreases opportunities to capture and remove N and P from stormwater runoff before it enters rivers, streams, and estuaries.

SECTION 7.32: NATURE, TIMING, AND MAGNITUDE OF NUTRIENT CHANGES

Stream water concentrations of both N and P tend to be higher in developed landscapes than in areas of lower population density, although higher concentrations can be found in streams in close proximity to specific agricultural and horticultural uses such as row crops and container nurseries (Paul & Meyer 2001). During nationwide monitoring of surface waters, USGS (2001) found that more than 70% of sampled urban streams exceeded minimum nutrient levels determined by USEPA to limit nuisance aquatic plant growth. Research conducted within and beyond the Pacific Northwest has documented increased P concentrations (up to five times beyond 'natural' or reference levels) associated with urban development (Wahl *et al.* 1997; Sonoda *et al.* 2001), impervious surface patch size (Carle *et al.* 2005), and wastewater effluent (Fisher *et al.* 2000; Coulter *et al.* 2004). Analysis of urban runoff of the Tualatin River basin (western Oregon) indicated that P concentrations in runoff usually exceed background P

⁴⁰ Ammonia (NH₃) is a form of nitrogen that can easily be lost from the soil through vaporization or leaching. In the soil, bacteria convert ammonia into nitrites (NO₂) which are toxic to plants. Other bacteria oxidize nitrites into nitrate (NO₃) which is readily used by plants. Nitrate is easily leached from the soil and can lead to eutrophication of surface waters. Total Kjeldahl N is the amount of nitrogen in organic substance determined by the Kjeldahl method of acid digestion. Total N is the amount of both organic and inorganic nitrogen.

⁴¹ Orthophosphate is an inorganic form of phosphorus readily available for plant uptake in the environment.

concentrations⁴² in tributaries of the Tualatin mainstem (ODEQ 2001b). In the Seattle (Washington) area, Brett *et al.* (2005b) found that streams draining urban dominated watersheds had, on average, 110% and 40% higher P and N concentrations (respectively) compared to streams draining forested watersheds. In Portland (Oregon), Hook & Yeakley (2005) found higher N concentrations in streams draining a small urban watershed than streams draining forested watersheds, but the urban stream concentrations were lower than streams in agricultural systems.

While the above studies have shown increased nutrient loads associated with developed areas, it is difficult to determine whether the increased nutrient loads are more attributable to higher inputs, reduced nutrient retention, or both. One approach is to quantify the degree to which nutrient availability increases in developed landscapes. For example, fertilizer application to residential landscaping constitutes a significant source of excess nutrients that varies considerably according to individual landowner preferences. Survey results reported by Nielson & Smith (2005) indicated that the majority (64%) of respondents in the Tualatin River basin (western Oregon) fertilize their lawns 2 to 3 times per year, a fertilizer frequency in excess of that advised by yard care specialists. However, the degree to which these nutrients are later transferred to streams is unknown.

The type of wastewater treatment used within a development (i.e., septic tank systems vs. municipal treatment facilities) also influences the magnitude of nutrient inputs to aquatic ecosystems. ODEQ (2001b) reported that total P levels in the Tualatin River (western Oregon) were greatly reduced when wastewater treatment plants increased their total P removal capabilities and implemented best management practices such as stormwater treatment, street sweeping, and educational programs. While this is encouraging, treated sewage effluent does constitute a significant nutrient source in other Pacific Northwest locations. For example, in western Washington, Inkpen & Embry (1998) reported that treated effluent contributes 22% of the N load in the Puyallup River and 15% of the P load in the Snohomish River. During summer drought seasons, municipal water discharge can constitute a significant fraction of instream flow that can result in seasonal increases in nutrient concentrations in affected streams (Brooks *et al.* 2006; Kaye *et al.* 2006).

Septic systems also contribute nutrients to ground and surface waters. In a comparison of developed North Carolina watersheds, Carle *et al.* (2005) found that developed lands outside city limits contributed higher total N and total P loads to streams than developed areas serviced by municipal wastewater treatment. Similarly, Hatt *et al.* (2004) found that septic tank densities strongly influenced N concentrations in streams of Melbourne, Australia. In Cape Cod (Massachusetts), Cole *et al.* (2006) used a stable isotopic form of N to determine the effect of land use on N loading in groundwater. Higher isotopic N concentrations found in urban watersheds were attributed to increased wastewaters from septic systems (Cole *et al.* 2006). In La Pine (Oregon), N isotope data indicated that septic tank effluent was the main source of nitrate in shallow groundwater in the area, whereas naturally occurring sedimentary organic matter was the main source of ammonia found in deep groundwater⁴³ (Hinkle *et al.* 2007). Steffy

⁴² Natural P loadings from groundwater in the Tualatin River basin are high and were taken into account in the background concentrations determined by ODEQ (2001c).

⁴³ Groundwater from the La Pine (Oregon) discharges into the Deschutes and the Little Deschutes Rivers but the amount of septic related nitrate entering surface waters was not determined in this study (Hinkle *et al.* 2007) or in others examining nutrient loading in the aquifer.

& Kilham (2004) analyzed stable isotopic forms of N to determine the degree to which aquatic food webs accumulated N derived from anthropogenic sources in developed landscapes. Stable isotope signatures revealed that for all trophic levels analyzed, N from anthropogenic sources was up to 10% greater than that observed in minimally disturbed systems. Sampling sites in close proximity to residential septic tank systems exhibited the highest anthropogenic N levels, leading Steffy & Kilham (2004) to conclude that improperly functioning septic systems contributed large amounts of anthropogenic N to aquatic ecosystems.

The distribution and number of landscape features that retain nutrients may also serve as important regulators of the nutrient loads delivered to surface waters. Developments can eliminate areas and create discrete areas that retain and remove significant quantities of N (i.e., denitrification). For example, stormwater detention ponds, ditches, gutters, and lawns all have the capacity to accumulate and retain water, N, and organic matter long enough for microorganisms to convert the forms of N typically used by aquatic plants to nitrogen gas (Kaye *et al.* 2006). Evidence supporting this contention comes from observations of stream N inputs that were lower than expected given the quantities of N accumulated on hydrologically linked impervious surfaces (e.g., Hope *et al.* 2004; Grimm *et al.* 2005) and high rates of denitrification in stormwater retention basins (Zhu *et al.* 2004). These processes, however, do not remove P although stormwater detention ponds may sequester it by adsorption to sediments and uptake by plants. While these features of developed landscapes may reduce or retain nutrients, it remains unknown to what degree these vegetated features will decrease nutrient inputs into Oregon waterways because most storm runoff occurs during the fall and winter when many plants become dormant (Grimm *et al.* 2008; Pickett *et al.* 2008).

Reduced nutrient retention within streams may play a significant role in increasing nutrient concentrations downstream of developments (Kaye *et al.* 2006). Stream ecosystems provide important connections among surrounding ecosystems including terrestrial uplands, groundwater, lakes, and downstream recipient ecosystems such as larger rivers, freshwater wetlands, and coastal estuaries. Nutrient retention within any given stream reach reduces nutrient transport to downstream ecosystems. Highly engineered stormwater drainage structures and altered channels accelerate downstream nutrient transport, reduce physical habitat heterogeneity that supports aquatic biota capable of direct nutrient uptake, disconnect channels from floodplains capable of sequestering nutrients, and therefore decrease in-stream nutrient retention (Kaye *et al.* 2006). Grimm *et al.* (2005) found that as stream structure declined in heavily urbanized areas of the arid southwestern US, longer stream lengths were required for biotic uptake of fixed amounts of N to occur. Similarly, Gibson & Meyer (2007) found that both N and P uptake rates in the Chattahoochee River (Georgia) were much lower than uptake rates measured in less heavily modified streams resulting in transport of anthropogenic nutrients many kilometers downstream. Gibson & Meyer (2007) also found that adsorption of P to suspended sediment temporally slowed downstream transport as sediments settled, but this effect was thought to be only temporary because sediment-bound P is readily mobilized during subsequent high-flow events.

In summary, the above evidence indicates that development not only increases the amount of nutrient available for delivery to streams, but also changes the degree to which streams and surrounding riparian and upland landscapes retain nutrients (Kaye *et al.* 2006). The comprehensive nature of these changes and their influences on nutrient loads in aquatic

ecosystems adjacent to or downstream of developed lands has received little research attention in the Pacific Northwest.

Key Findings: Nutrients

- Stream water concentrations of both nitrogen and phosphorus tend to be higher in developed landscapes than in areas of lower population density.
- Development not only increases the amount of nutrients available for delivery to streams, but also changes the degree to which streams and surrounding riparian and upland landscapes retain nutrients.
- It is difficult to determine whether increased nutrient loads observed in aquatic ecosystems affected by development are more attributable to higher inputs or to reduced retention, although both factors clearly contribute.
- During summer drought seasons, municipal water discharge can constitute a significant fraction of instream flow that can result in seasonal increases in nutrient concentrations in affected streams.

Section 7.4: Water Temperature in Urban and Rural-residential Areas

In this section, IMST describes the factors and mechanisms that alter water temperature and the nature, timing, and magnitude of water temperature changes in streams, rivers, and estuaries affected by development.

SECTION 7.41: FACTORS AND MECHANISMS THAT ALTER WATER TEMPERATURE

Salmonids require relatively cold water during most life history stages and can experience both lethal and sublethal effects from elevated water temperature (reviewed in McCullough 1999, McCullough *et al.* 2001; IMST 2004; Table 7-1). Temperature governs both development rate and survival of salmonid eggs and alevins (Murray & McPhail 1988; McCullough *et al.* 2001; Nelitz *et al.* 2007). Spawning activity (i.e., redd excavation, egg deposition, and fertilization) may cease if water temperatures are not favorable (McCullough *et al.* 2001; Sauter *et al.* 2001). Temperature cues often initiate seaward migrations of juvenile salmonids (Roper & Scarnecchia 1999; Achord *et al.* 2007), as well as the return of sexually mature adults (McCullough *et al.* 2001; Sauter *et al.* 2001). Altered temperatures can interfere with adult migration by advancing or delaying it (Quinn & Adams 1996; Cooke *et al.* 2004; Goniea *et al.* 2006), by creating thermal barriers to upstream movement (Alabaster 1988; Quinn *et al.* 1997), and by inducing stress-related infection and mortality (Macdonald *et al.* 2000; Rand *et al.* 2006; Newell *et al.* 2007). Other temperature-mediated mechanisms that affect salmonid growth and survival include disease and parasite resistance, competitive ability, and predation risk (Brett 1956; Poole *et al.* 2001, 2004; Nelitz *et al.* 2007). Given the mechanistic links between water temperature and salmonid health, it is not surprising that temperature is a key determinant of salmonid distribution and abundance throughout the Pacific Northwest, at both river reach (Nielsen *et al.*

1994; Tiffan *et al.* 2006) and river basin scales (Li *et al.* 1994; Torgersen *et al.* 1999; Ebersole *et al.* 2001; Sauter *et al.* 2001).

Table 7-1. Estimates of temperature ranges at or above which various life-history stages of Pacific Northwest salmon and bull trout are likely to experience adverse effects. Ranges include important variation within and among salmon species. Most salmonids are commonly observed at summer habitat temperatures ranging from 50°F to 63°F (10°C to 17°C), with the exception of bull trout that are more commonly observed at summer habitat temperatures ranging from 43°F to 54°F (6°C to 12°C). Table adapted from Poole *et al.* (2001).

Life-History Stage	Temperature Response	Anadromous Salmon Temperature Range	Bull Trout Temperature Range
Adult Migration	Blocked	70-72°F (>21-22°C)	N/A
Adult Migration	Cued	N/A	50-55°F (10-13°C)
Spawning	Initiated	45-57°F (7-14°C)	<48°F (<9°C)
Egg Incubation	Optimal	43-50°F (6-10°C)	36-43°F (2-6°C)
Smoltification	Suppressed	>52-59°F (>11-15°C)	N/A
Adult Survival	Lethal (one week exposure)	>70-72°F (>21-22°C)	N/A
Juvenile Survival	Lethal (one week exposure)	>73-75°F (>23-24°C)	>72-73°F (>22-23°C)

The pathways by which rural-residential and urban land uses influence stream temperatures are complex and often interconnected making it difficult to determine the importance of individual factors (Figure 7-3; IMST 2004). Water temperature is a function of heat load, stream discharge, and water volume. Therefore, streams become more susceptible to temperature changes when flows and/or volumes are low and/or heat energy sources are abundant (Poole & Berman 2001; IMST 2004). In the Pacific Northwest, stream temperatures can naturally exceed those optimal for salmonid spawning and rearing, particularly during periods of summer drought (Poole *et al.* 2001, 2004). Under these conditions, additional warming caused by anthropogenic effects can be particularly stressful for salmonids and other biota adapted to cold water.

A diverse set of mechanisms have the potential to increase heat inputs to streams and rivers in rural-residential and urban areas. Industrial and municipal facilities discharge significant volumes of heated wastewater into Oregon's streams and rivers on an annual basis (see Section 3.0 of this report). Constructed surfaces such as pavement and rooftops absorb more heat than vegetated surfaces and can absorb enough heat to alter the climate in developed landscapes, a phenomenon termed "heat island effect" (e.g., Voogt & Oke 2003; Arnfield 2003; Souch & Grimmond 2006). Water flowing over these heated surfaces picks up excess heat and can become a significant non-point source of thermal pollution when routed to streams (Nelson & Palmer 2007). Because of the low levels of rainfall that occur during summer months in Oregon, it is unclear whether the heating of storm flows over impervious areas in Oregon cities

significantly affects stream water temperatures. Detention ponds, designed to regulate the rate and volume of stormwater flow through developed areas also increase stream temperature (Galli 1990). These individual heat sources influence temperatures immediately downstream to varying extents and contribute to the cumulative heat load received by streams. Dams and reservoirs associated with municipal drinking water supplies can also alter the temperatures of streams outside urban

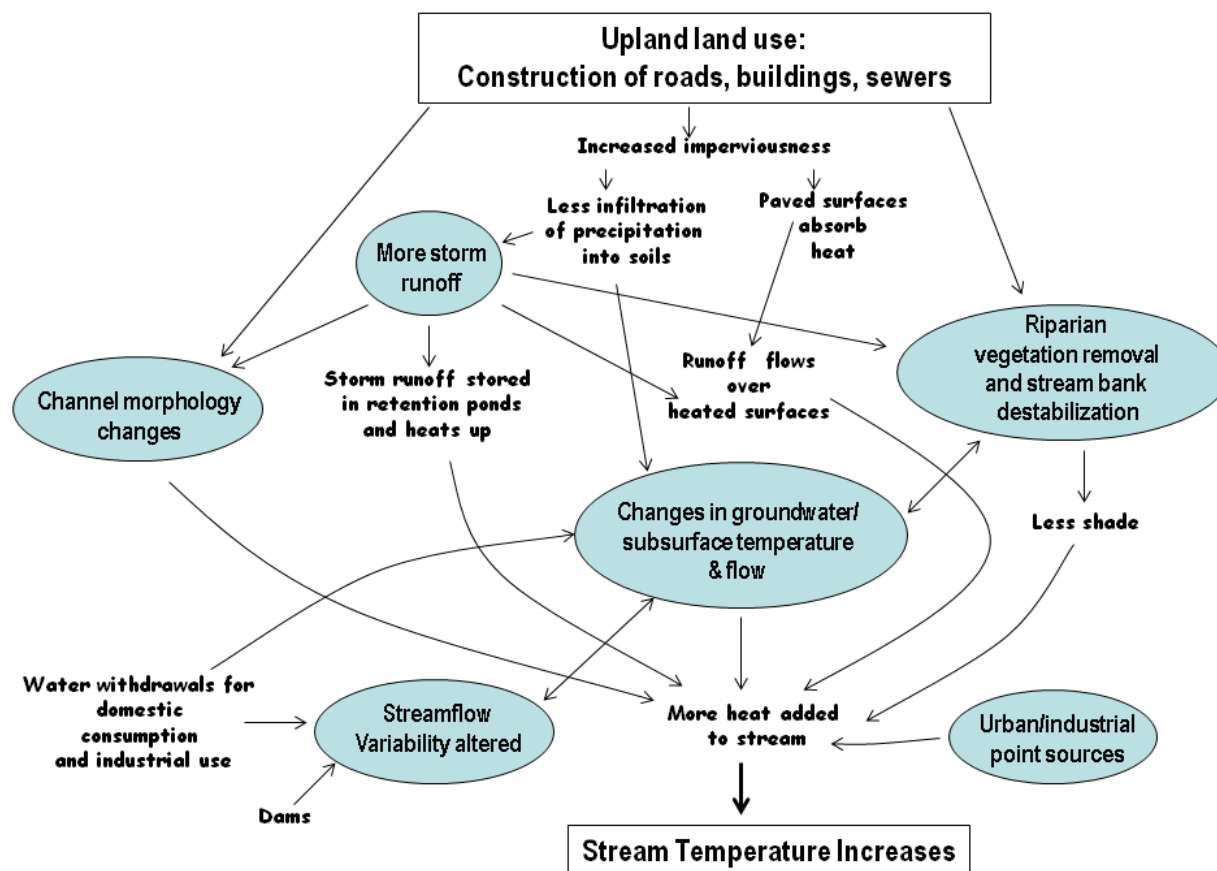


Figure 7-3. Hydrologic and physical habitat factors associated with developed landscapes that influence stream temperature variability. Habitat factors are shown in shaded ellipses. For detailed information on these influences, see IMST (2004).

growth boundaries. For example, dams on the Bull Run River, constructed to regulate water supply to the Portland metropolitan area, release water that is cooler than unregulated temperatures in early summer and warmer than ambient temperatures in late summer (ODEQ 2005; NMFS 2006b).

Riparian vegetation can directly affect stream temperatures by intercepting short-wave radiation during the day and insulating the stream from long-wave radiation loss at night. Through evapotranspiration, vegetation reduces air temperatures by converting sensible heat to latent heat (McPherson 1994). In addition, riparian vegetation also partially controls channel morphology,

streamflow, groundwater connectivity, infiltration and percolation rates, wind speed, humidity, and soil temperature, all of which influence stream temperature (reviewed in IMST 2004). As a result, removal of riparian vegetation, a common consequence of development (Ozawa & Yeakley 2007), influences stream temperatures via numerous mechanisms.

Channel morphology, particularly width and depth, influences the amount of heat gained or lost from a stream (reviewed in Poole & Berman 2001; IMST 2004). Wider channels have more stream surface area available to exchange radiant and atmospheric heat energy and are less effectively cooled by riparian forests. As channel depth decreases, solar radiation penetrates a larger fraction of the water volume and influences heating and cooling rates (Nelson & Palmer 2007). As watersheds undergo development, channel dimensions often change in ways that lead to increased stream temperatures.

Groundwater contributions to flow are an important factor in determining stream temperature (reviewed in Poole & Berman 2001). Groundwater, insulated from daily and seasonal temperature fluctuations, exhibits relatively stable temperatures that buffer stream temperatures when groundwater constitutes a large proportion of flow. Groundwater influences on surface water temperatures are particularly strong near their input site. Groundwater can also form pockets of cold water that create reach-scale thermal heterogeneity (reviewed in Jones & Mulholland 2000) and constitute important habitat features for cold-water biota including salmonids (Poole *et al.* 2001, 2004; Ebersole *et al.* 2003). Depending on watershed geology and stream channel structure, two sources of groundwater can influence stream temperature (Poole & Berman 2001). Water flowing along shallow pathways below river channels and riparian zones (referred to as hyporheic⁴⁴ groundwater) exchanges with surface waters over relatively short timeframes (e.g., minutes to months). Water stored in deep aquifers (phreatic groundwater) also exchanges with surface flows but typically over longer intervals.

Hydrologic alterations that reduce groundwater discharge to streams can increase stream temperature and reduce reach-scale temperature heterogeneity, thus reducing habitat available to aquatic organisms that require cooler temperatures (reviewed in Poole & Berman 2001; IMST 2004). Movement of hyporheic groundwater is a function of streamflow variability, stream channel pattern, and streambed complexity (reviewed in Poole & Berman 2001), all of which are altered by development. In Oregon, high flows and floods occur during the winter and spring months when water temperatures are at their coldest. Recharge of hyporheic groundwater during these periods may create an important source of cold water that buffers stream temperatures during sensitive baseflow periods (Poole & Berman 2001). River channelization confines flow and limits river-floodplain interactions that recharge shallow groundwater sources, thus reducing groundwater discharge during baseflow. Channel modifications such as straightening, diking, dredging, and armoring focus stream energy toward the center of the channel further reducing channel heterogeneity, leading to channel incision and, disruption of hyporheic flow when it is present (Hancock 2002). Removing large wood also decreases streambed complexity and can reduce hyporheic groundwater exchange in some streams.

Disturbance and removal of upland vegetation typically decreases infiltration of precipitation inputs through the soil, limits deep aquifer recharge, increases delivery of fine sediments that clog channel substrates and restrict hyporheic exchange, and alters the timing, volume, and

⁴⁴ The hyporheic zone is the region in streambed and bank sediments where exchange between surface and ground waters may occur.

magnitude of peak flows that shape channel morphology (see Section 4.0 of this report). When reduced infiltration and percolation limits groundwater aquifer recharge, reduced summer baseflows can lead to elevated stream temperatures in some urban streams (Finkenbine *et al.* 2000).

Municipal water withdrawals from both shallow and deep groundwater sources reduce flow in nearby streams and rivers, thus concentrating heat energy and increasing stream temperature. If the withdrawn water is returned to the stream, it is likely in the form of heated industrial or wastewater effluent and adds to the stream's overall heat load (Kinouchi 2007; Kinouchi *et al.* 2007). Approximately 70% of Oregon residents, including 90% of residents in rural developments, use groundwater as their primary water source (Bastasch 2006). Human population growth will certainly increase the demands made on Oregon's groundwater resources, particularly under the constraints imposed by a changing climate (Houston *et al.* 2003). Increasing use of groundwater resources has the potential to reduce streamflows and increase stream temperatures (Dole & Niemi 2004; ISAB 2007a, b; See Section 4.2 of this report).

SECTION 7.42: NATURE, TIMING, AND MAGNITUDE OF WATER TEMPERATURE CHANGES

Hydrologic and physical habitat changes, as well as point and non-point pollution sources that accompany development, can alter the timing and amount of heat energy and/or water delivered to aquatic habitats and strongly influence temperature fluctuations in these environments (Figure 7-3; Poole & Berman 2001). Stream temperature changes include increases in the maximum and decreases in the minimum daily and seasonal temperatures (Poole & Berman 2001). The likelihood that environmental changes associated with development will lead to stream temperature changes varies depending on reach- to basin-scale characteristics including riparian vegetation context, landscape position (e.g., headwaters vs. mainstem), stream size, and watershed geology.

Compared to undeveloped watersheds, research focusing on changes in stream temperatures in rural-residential and urban streams is limited (Krause *et al.* 2004; Nelson & Palmer 2007). Models used to predict the stream temperature response to development provide valuable information on the how altered physical habitat and hydrologic regimes affect stream temperature, and highlight the importance of riparian shade and channel width (e.g., LeBlanc *et al.* 1997; Krause *et al.* 2004; Nelson & Palmer 2007). However, interpreting conclusions drawn from these modeling efforts requires caution because the models do not include important point source thermal discharges (e.g., stormwater sewers or wastewater treatment effluent) known to increase stream temperature in developed landscapes.

LeBlanc *et al.* (1997) identified riparian shading, groundwater discharge, and stream width as variables that have the potential to influence stream temperature. In general, LeBlanc *et al.* found that stream temperature increased as riparian canopy density decreased and channel width-to-depth ratios increased, but results varied depending on stream orientation on the landscape (e.g., north to south). The magnitude of temperature change increased when development also reduced base flow discharge and groundwater exchange with the stream (LeBlanc *et al.* 1997). Krause *et al.* (2004) found that the combined effects of reduced riparian shade, increased channel width, and increased stormwater runoff temperature associated with high-density development produced the largest changes in mean daily stream water temperature. Nelson & Palmer (2007) integrated empirical relationships between land use and stream temperature into a model used to

predict stream temperature change in a rapidly urbanizing watershed. The proportion of deforested land, particularly within riparian areas, was identified as an important variable explaining increased temperature.

In areas that receive substantial spring and summer precipitation, stormwater flow over heated pavement may be a source of heat to streams. For example, Wang *et al.* (2003b) observed a positive linear relationship between summer maximum water temperature and the amount of connected impervious area across 39 sites in Wisconsin and Minnesota. In Maryland, Nelson & Palmer (2007) found that high runoff events generated by localized summer storms caused surges in stream temperature (+ 3.5°C on average but ranged to > 7°C) that persisted for up to 3 hours. In contrast, Booth *et al.* (2001) found that urbanization caused only minor elevation of summer stream temperature in King and Snohomish counties (western Washington) and attributed this result to the combined effects of groundwater baseflow sources and infrequent summer precipitation. IMST is unaware of any studies that examined whether runoff passing over paved surfaces contributes significantly to stream heating in Oregon basins. Little precipitation falls in Oregon when air temperatures are high, indicating that results may be similar those of Booth *et al.* (2001); however, future changes in climate may alter this precipitation pattern (Hamlet & Lettenmaier 1999; Leung & Wigmosta 1999; Miles *et al.* 2000; Mote *et al.* 2003; Payne *et al.* 2004; Claessens *et al.* 2006).

Research carried out in Japan demonstrated that wastewater from municipal treatment plants can constitute a large proportion of the anthropogenic heat delivered to streams and can drive significant stream temperature increases over extended periods of urban development (Kinouchi *et al.* 2007). As the number and density of residences increase in a development, both the volume and temperature of wastewater generated by residential use can increase. Kinouchi (2007) demonstrated that increases in temperature and volume of water generated by residential uses increased the annual mean temperature of wastewater generated by municipal treatment plants in Tokyo, Japan by 5.5°C over four decades. In general, the effects of municipal treatment plant wastewater on stream temperature depend on the temperature and volume of wastewater discharged and on the flows and volumes of the receiving stream.

A few studies have documented the cumulative magnitude of water temperature change resulting from rural-residential or urban development. Scientists evaluated the water quality of six urban areas across Oregon using ODEQ's Water Quality Index (Figure 7-1; Oregon Progress Board 2000)⁴⁵. In Medford (southwest Oregon), Eugene-Springfield (western Oregon), Bend (central Oregon), and La Grande (eastern Oregon) water temperature ranked "poor". Portland (western Oregon) temperature trends were good or unclear. Temperature trends were potentially declining in quality in many western Oregon watersheds including the Clackamas River, Fanno Creek, Willamette River, and Columbia Slough (Oregon Progress Board 2000). Water quality data collected by the City of Portland (2004) and ODEQ (2001b) indicate that the cumulative effects of development both increase heat loading and reduce flow in urban streams, resulting in summer stream temperatures that often exceed those required by salmonids (Table 7-1).

When a water body in Oregon fails to meet state water temperature standards, ODEQ performs a thorough analysis of point and nonpoint sources of increased heat loading as part of a TMDL

⁴⁵ Bend, Coos Bay, Eugene-Springfield, La Grande, Medford, and Portland Metropolitan Area. These cities were selected because they represent different ecoregions, sizes, densities, and socio-economic populations.

analysis⁴⁶. In the Tualatin River watershed (western Oregon), approximately 12% of the land is used for urban and rural-residential developments (ODEQ 2001b). Stream temperature modeling for this watershed, carried out as part of the TMDL process, has indicated that with minimal anthropogenic heat sources, 98% of the stream network would be below a maximum daily temperature threshold of 64°F (17.8°C)⁴⁷ (ODEQ 2001b). Of the total heat loading that occurs during critical summer months, 41% is derived from natural background sources, 52% from anthropogenic non-point sources and 7% from point sources. Fifteen industrial and municipal facilities permitted to discharge heated effluents into the river contribute the 7% of total summertime heat loading attributed to point sources. The temperatures of effluent discharged by individual facilities ranges from 66°F (19°C) to 88°F (31°C). Effluents discharged by two wastewater treatment facilities (Durham and Rock Creek) contribute disproportionately to the point source heat loading for the Tualatin River. For example, the Rock Creek Wastewater Treatment Plant discharged water at 72°F (22°C)⁴⁸ into a reach with an average temperature of 65°F (18°C). The discharge increased the average river temperature 1.5°F (0.83°C) for almost a mile downstream (ODEQ 2001b).

ODEQ (2001b) attributed non-point source anthropogenic heat loading (52% of the total heat loading) to increased solar radiation along stream reaches where riparian vegetation has been disturbed or removed, thus reducing stream shading. Solar radiation loads derived from anthropogenic disturbance ranged from 44% along the mainstem Tualatin River (Oregon) to between 88% and 99% in Tualatin tributaries. These data mirror findings by Booth *et al.* (2001) that changes to the riparian canopy have the most direct influence on summertime stream temperatures in the heavily developed Puget Sound (Washington) lowlands. These data also corroborate modeling results that identify riparian condition as a key mechanism in the stream temperature response to development (LeBlanc *et al.* 1997; Krause *et al.* 2004; Nelson & Palmer 2007). Riparian vegetation condition is highly correlated with a suite of variables that collectively regulate stream temperature including stream bank erosion, channel stability, and channel width. Therefore, it is not surprising that riparian condition and increased solar radiation exposure are frequently identified as important factors in the stream temperature responses to development.

Key Findings: Stream temperature

- Rural-residential and urban land uses can influence stream temperature by increasing effluents discharged from industrial and municipal facilities and detention ponds, dams, reservoirs and water withdrawals associated with municipal drinking water supplies.
- Rural-residential and urban land uses can directly and indirectly influence stream temperature by removing riparian vegetation, modifying channel morphology, and reducing groundwater discharge.

⁴⁶ For an additional example of the TMDL process in Oregon see the ODEQ (2007) factsheet *Reducing Temperature in the Willamette Basin* at http://www.deq.state.or.us/wq/pubs/factsheets/willamette/07wq000_temperature.pdf. Accessed May 25, 2010.

⁴⁷ Tualatin subbasin temperature standards (ODEQ 2001c) mandate that no measurable surface water increase resulting from anthropogenic activities is allowed in salmonid rearing streams where water temperatures exceed 17.8°C (64°F) or salmonid spawning streams where water temperatures exceed 12.8°C (55°F).

⁴⁸ Measurements taken July 27, 1999 (ODEQ 2001c).

Section 7.5: Dissolved Oxygen in Urban and Rural-residential Areas

In this section, IMST describes the factors and mechanisms that alter dissolved oxygen concentrations and the nature, timing, and magnitude of dissolved oxygen changes in streams, rivers, and estuaries affected by development.

SECTION 7.51: FACTORS AND MECHANISMS THAT ALTER DISSOLVED OXYGEN CONCENTRATIONS

Dissolved oxygen (DO) dynamics are a major determinant of aquatic community species composition. All salmonids and many other fish species require more than 5 mg O₂/L to survive. The amount of DO present in a body of water reflects a balance among:

- Oxygen production via photosynthesis;
- Diffusion of oxygen between water and the atmosphere;
- Oxygen solubility that decreases with increasing temperature and elevation;
- Chemical reactions that require oxygen creating a chemical oxygen demand; and
- Oxygen consumed by respiration of plants and animals including decomposition of organic matter by microorganisms that creates biological oxygen demand.

Dissolved oxygen dynamics in aquatic ecosystems are complex and depend on interactions among these physical, chemical, and biological processes. Mechanisms that alter DO concentrations in rural-residential and urban streams include water temperature changes, effluents from municipal and industrial wastewater-treatment facilities, leaks and overflows from sewage lines and septic tanks, stormwater runoff, and decaying aquatic plants that have undergone rapid biomass production due to excess nutrient availability. Water temperature can affect DO in two ways. First, higher water temperatures reduce the amount of oxygen that can remain dissolved in the water⁴⁹. Second, the rates of biological and chemical processes consuming oxygen increase with higher temperatures. Metabolic rates of aquatic organisms also increase as temperatures rise, thus increasing their oxygen requirements during periods of decreased oxygen solubility and availability (Brett 1964; Matthews & Berg 1997).

Biological processes strongly influence DO concentrations. Photosynthesis increases the supply of DO and consumes carbon dioxide and bicarbonate. Respiration and decomposition of plant biomass decrease DO concentrations and release carbon dioxide. When elevated nutrient concentrations support excessive growth of algae and other aquatic plants, daytime photosynthesis and nighttime respiration of plants can cause DO concentrations to have larger diurnal fluctuations (Smith *et al.* 1993; Gordon *et al.* 2005). In many water bodies, DO concentrations may not meet water quality standards (see ODEQ 2001b, 2005 for examples) because of excessive algae growth and respiration. In these streams, DO concentrations supersaturate during the day and rapidly decline at night. As a result, ODEQ monitors chlorophyll *a* concentrations (desired is <15µg/L) to determine when algal growth may cause DO problems (ODEQ 2001b).

⁴⁹ For example, the maximum DO concentration (100% saturation) in freshwater at sea level is 14.6 mg O₂/L at 0°C (32°F), but the maximum DO concentration under identical conditions at 20°C (68°F) is only 9.1 mg O₂/L (Colt 1984).

Aquatic microorganisms can decompose organic solids in the water column or on the bottom of streams. Sources of organic matter vary and include natural sources such as leaf litter, or anthropogenic sources such as stormwater runoff, effluent from wastewater treatment facilities, algal detritus from algae blooms stimulated by excess nutrient availability, and soil erosion (ODEQ 2001b). For example, in some cities, sewer networks allow stormwater runoff already loaded with pollution from urban surfaces to mix with large volumes of urban wastewater during storm events. This mixture of stormwater and raw sewage can be discharged directly into streams and rivers when storm flows exceed the sewer network capacity.

While Oregon cities have greatly reduced the frequency of wastewater overflow events (see Section 3.0 of this report), they still occur. The organic matter that enters rivers during combined sewer overflows can dramatically increase the amount of oxygen consumed by microorganisms (i.e., biological oxygen demand) downstream of the plume, thus depleting DO available to aquatic biota as long as the organic material remains (Even *et al.* 2004). Microorganisms also convert ammonia (originating in wastewater overflows or treatment plant effluent; ODEQ 2001b) to nitrite and nitrate (referred to as nitrification), a biological process that requires oxygen.

Compared to decomposition in the water column and nitrification, decomposing sediments may remain a DO sink for much longer periods after pollution discharges cease because sediments that settle out of the water column decompose slowly. Consequently, organic-containing sediments delivered by stormwater runoff or combined sewer overflows may trigger DO deficiencies long after the rain event that delivered the sediment to the river has ended.

Some toxic substances can reduce DO concentrations in streams. The USEPA estimated that 40 million liters of aircraft deicing and anti-icing fluids are discharged annually to receiving waters in the US (USEPA 2000c). Portland International Airport (Oregon) has released deicing fluids into the Columbia Slough and Columbia River but has also made efforts to better contain this runoff (GAO 2000). Glycols, which constitute the majority of aircraft deicing formulations, have high chemical oxygen demand and can cause significant DO reductions when they enter streams through stormwater runoff (Corsi *et al.* 2006). Low DO levels can also alter water chemistry in ways that accelerate the release of phosphorus and toxic chemicals, such as heavy metals, from sediments and prevent the detoxification of ammonia (a substance directly toxic to aquatic organisms) by oxygen requiring microorganisms.

SECTION 7.52: NATURE, TIMING, AND MAGNITUDE OF DISSOLVED OXYGEN CHANGES

Wang *et al.* (2003a) compared in-stream metabolic activities between an urban and an agricultural stream and found that the urban stream was always heterotrophic (consuming DO) compared to the agricultural stream that was periodically autotrophic (producing oxygen). Their research did not identify the likely causes of this difference. Similarly, Rodriguez *et al.* (2007) identified an inverse relationship between DO in estuaries of the northeastern US and total urban area.

Urban stormwater runoff and combined sewer overflows are common sources of low DO concentrations and high rates of oxygen consumption. Water quality monitoring at 83 sites distributed across the US regularly recorded depleted DO concentrations in urban areas, particularly during periods of wet weather (Keefer *et al.* 1979). Heaney *et al.* (1980) reviewed research results on DO downstream of urban areas and concluded that approximately one-third of the sites exhibited lowest DO concentrations after storms. Pitt (1979) also demonstrated that

stormflows to streams and rivers increased biological and chemical oxygen consumption rates (i.e., biological and chemical oxygen demand). Impacts of urban stormwater can last much longer than the duration of a single storm. Pitt (1995) later documented a lag between rainfall and peak oxygen demand; 10 to 20 days after a storm event, oxygen demand increased to levels 5- to 10-fold greater than those observed in the initial 1 to 5 days after the storm event.

Low DO concentrations in the mainstem Tualatin River (Oregon) and many of its tributaries have led ODEQ to list them as water quality impaired (ODEQ 2001b). Unacceptable DO levels typically occur during the late summer and early fall months and result from increased temperature, low flows, algal detritus, and high oxygen demand of river sediments. In most cases, the oxygen sink created by sediments, exacerbated by increased temperatures, is a significant contributor to oxygen depletion.

Key Findings: Dissolved oxygen

- Mechanisms that alter dissolved oxygen concentrations in rural-residential and urban streams include water temperature changes, effluents from municipal and industrial wastewater-treatment facilities, leaks and overflows from sewage lines and septic tanks, stormwater runoff, and decaying aquatic plants that have undergone rapid biomass production due to excess nutrient availability.
- Urban stormwater runoff and combined sewer overflows are common sources of low oxygen concentrations and high rates of oxygen consumption. Unacceptable DO levels typically occur during the late summer and early fall months and result from increased temperature, low flows, algal detritus, and high oxygen demand of river sediments.

Section 7.6: Toxic Pollution in Urban and Rural-residential Areas

In this section, IMST describes the types, origins, and potential impacts on aquatic biota of anthropogenic-derived toxic chemicals entering Oregon's waterways. Toxic contaminants create complex problems for aquatic organisms and have important implications for natural resource managers. Because of this, the following section on toxic contaminants departs from the format of previous report sections so that biotic responses to toxic contaminants can be presented in greater detail than the general responses discussed later in this report.

SECTION 7.61: ORIGINS, PREVALENCE AND DIVERSITY OF CONTAMINANTS ASSOCIATED WITH DEVELOPMENT

Some contaminants originate from a few significant sources. In the US, widespread use of the volatile organic compound methyl *tert*-butyl ether (MTBE) as a gasoline additive contaminated ground and surface waters in many communities, stimulating legislation to restrict or ban its use in several states⁵⁰ (and potentially nationwide⁵¹; Hamilton *et al.* 2004; Moran *et al.* 2005). Atmospheric deposition of coal combustion and industrial waste incineration products are a major source of mercury contamination in many ecosystems (Hamilton *et al.* 2004; Peterson *et al.* 2007).

Other pollutants originate from numerous low-level sources that make significant contributions when considered in total. Copper is a common pollutant in urban stormwater runoff and can originate from some building materials, wood preservatives, pesticides, and vehicle brake pads (Beasley & Kneale 2002). Several pesticides are widely used to maintain landscaping, therefore numerous public or privately owned land parcels can act as sources for these water contaminants. Herbicides such as 2,4-D and glyphosate were detected in a large proportion of water samples collected between 2000 and 2005 in the Clackamas River basin (western Oregon) (Carpenter *et al.* 2008).

Legacy compounds are substances that were manufactured and in use for years or decades and are now banned, but continue to pollute Oregon's aquatic ecosystems (Wentz *et al.* 1998; Ebbert *et al.* 2000; ISAB 2007b; LCREP 2007). The manufacture and use of polychlorinated biphenyls (PCBs) in the US was banned in 1979 but these materials still occur in streams and estuaries (King *et al.* 2004), including those associated with urban areas along the Willamette River (Black *et al.* 2000). King *et al.* (2004) concluded that urban areas bordering the Chesapeake Bay may still contain active sources of PCBs and pointed to the need for further research into historical and contemporary storage and delivery of these chemicals to surface waters and sediments.

Every year, new compounds are approved by regulatory agencies for use in the US. Guidelines for general water quality or aquatic life criteria are not required for many of these compounds prior to their approval for use (Ebbert *et al.* 2000; LCREP 2007; Carpenter *et al.* 2008). For example, more than 100 point sources legally discharge effluent containing unregulated contaminants directly into the lower Columbia River estuary (LCREP 2007). Up to 200 different

⁵⁰ June 2004 USEPA bulletin EPA-420-B-04-009 titled *State Actions Banning MTBE (Statewide)*. <http://www.epa.gov/mtbe/420b04009.pdf>. Accessed on October 7, 2010.

⁵¹ March 2004 USEPA bulletin EPA420-F-00-010 titled *Control of MTBE in Gasoline* <http://www.epa.gov/otaq/consumer/fuels/mtbe/f00010.htm>. Accessed on October 7, 2010.

chemicals have been identified in wastewater treatment plant effluent and combined sewer overflows (Ritter *et al.* 2002). The paucity of information regarding the occurrence of these contaminants in the environment or their effects on aquatic organisms clouds our understanding of their effects on aquatic ecosystems. Several sources identify compounds that fall into this information gap as *emerging contaminants* (e.g., Ritter *et al.* 2002; Brooks *et al.* 2006; Ellis 2006; ISAB 2007b; LCREP 2007). Emerging contaminants are not always newly created or approved compounds. This term also applies to substances recently recognized as unregulated and potentially significant water pollutants, even though they may have been present in runoff and wastewater for decades. For example, pharmaceuticals, some of which act as endocrine disruptors⁵², have been discharged into surface and ground waters for many years but efforts to survey and study them as pollutants have only recently begun (Ritter *et al.* 2002). It is the perception of the IMST that many, if not most, of the potentially toxic substances are not included in water quality analyses. This presents a further difficulty in assessing urban contaminants in surface and ground waters.

Emerging contaminants identified by various authors include the following examples.

- 1) There may be as many as 6 million different *pharmaceuticals and personal care products*, collectively known as PPCPs, commercially available worldwide. The development of new PPCPs and use of existing PPCPs increases annually (Kolpin *et al.* 2002; Ritter *et al.* 2002; Cui *et al.* 2006; Brooks *et al.* 2006; Ellis 2006). Examples of human and veterinary pharmaceuticals found in wastewater effluent include analgesics such as ibuprofen, antibiotics, anti-inflammatories, psychiatric drugs, lipid regulators, beta-blockers, x-ray contrast media, steroid estrogens and other hormones, and miscellaneous chemicals such as caffeine. Examples of personal care products include antiseptics, detergents, antibiotics, bioactive food supplements, cosmetics, fragrances, insect repellent, and sunscreen.

These substances primarily enter aquatic ecosystems when treated wastewater is discharged into receiving waters (Ritter *et al.* 2002; Ellis 2006). Synthetic estrogen originating from birth control pills, for example, is essentially untouched by sewage treatment (Parrott & Blunt 2005). Few generalizations can be made regarding modes of action or risk to aquatic organisms because these compounds range widely in chemical structure. While the environmental effects of many PPCPs have not been studied, there is a growing body of evidence indicating that some classes of substances (e.g., estrogenic compounds) pose significant threats to aquatic organisms (Santos *et al.* 2010).

- 2) Polybrominated-diphenyl-ethers (PBDEs) are widely used as flame retardants in various textiles, plastics, foams, and fire extinguishers. PBDEs leach from the plastics, electronics, and textiles in which they are used (de Wit *et al.* 2002). These compounds may enter surface and ground waters through treated wastewater and septic field effluent (Rayne *et al.* 2003; Elliott *et al.* 2005; ISAB 2007b; LCREP 2007). Another source of environmental contamination may be industrial facilities that either manufacture PBDEs and other flame retardants or use them in the manufacturing of other consumer products (de Wit *et al.* 2002). PDBEs have been isolated from tissues of aquatic organisms and their terrestrial predators residing in urbanized estuaries with increasing incidence and at

⁵² Endocrine disruptors are substances foreign to an organism that act like hormones and that stop the production or block the transmission of hormones in the body.

higher concentrations. Rayne *et al.* (2003) sampled mountain whitefish (*Prosopium williamsoni*) from portions of the Columbia and Kootenya Rivers (British Columbia, Canada) between 1992 and 2000 and documented a 12-fold increase in PBDE concentrations in whitefish tissues over this period. PBDEs are similar to PCBs in terms of their environmental persistence, chemical behavior, and immunotoxicity (Table 7-2; de Wit 2002; Morace 2006; LCREP 2007). Concern over public health and environmental effects of these contaminants caused both the Washington⁵³ and Oregon⁵⁴ state legislatures to ban some PBDEs beginning in 2011.

- 3) Some forms of polycyclic aromatic hydrocarbons (PAHs) are considered emerging contaminants. PAHs comprise a large group of natural and anthropogenic organic compounds and are present in creosote, asphalt, soot, roofing tar, crude oil, gasoline, and diesel fuel (Reynaud & Deschaux 2006; LCREP 2007). Anthropogenic PAHs probably enter surface waters via creosote treated structures, industrial effluent, episodic fuel spills, and runoff containing petroleum products and sediment-bound PAHs that have accumulated on impervious surfaces. PAHs arise from numerous sources in developed landscapes and are considered to be ubiquitous in aquatic ecosystems (Meador *et al.* 1995; van Metre *et al.* 2000). However, the transport, fate, and effects of PAHs in aquatic ecosystems requires additional research.
- 4) Nanomaterials are a diverse group of chemical compounds smaller than 100 nm in size that exhibit unique, size-dependent properties (Guzmán *et al.* 2006). It is anticipated that nanomaterials will be widely used in many applications in technology and industry (Dahl *et al.* 2007). There is concern with potential environmental impacts of nanomaterials that recently have become more ubiquitously manufactured and distributed, including their potential for toxicity to fish (Teuten *et al.* 2009). Laboratory-based experiments have demonstrated toxicity of several nanomaterial compounds in vertebrate organisms (reviewed by Guzmán *et al.* 2006). The behavior of manufactured nanomaterials in natural environments has been raised as a concern for this rapidly developing technology (Guzmán *et al.* 2006; Dahl *et al.* 2007). Limited research is currently underway to determine the potential for nanomaterials to become environmental contaminants (e.g., Harper *et al.* 2008).
- 5) Several chemicals used in the manufacture of plastics and personal care products have received increasing attention in surveys for water contaminants. Two examples of such contaminants include bisphenol A and a class of compounds called phthalates. Otherwise known as plasticizers, these chemicals are used to manufacture plastic products such as toys, food containers, medical devices, and vinyl (particularly polyvinyl chloride or PVC) used in flooring and wall coverings and similar products (see Oehlmann *et al.* 2009; Clara *et al.* 2010; and references cited therein). Phthalates are also used to carry fragrances and modify the consistency of personal care products such as nail polish (OEC 2007; Oehlmann *et al.* 2009). Because plastics have become a ubiquitous feature of human societies, plastic debris continuously enters natural ecosystems, is disposed of in landfills, or simply remains in use (Barnes *et al.* 2009). Plasticizers can enter aquatic

⁵³ <http://www.watotoxics.org/issues/pbde/legislative-update>. Accessed on July 24, 2008. For a summary of proposed and passed legislation related to PBDE bans in the US and elsewhere, see <http://www.ewg.org/node/26976>. Accessed on September 18, 2008.

⁵⁴ <http://www.leg.state.or.us/09reg/measpdf/sb0500.dir/sb0596.en.pdf>. Accessed May 4, 2010.

ecosystems via waste water effluent generated during manufacturing processes or by leaching directly from finished products that contain them (Oehlmann *et al.* 2009; Teuten *et al.* 2009; Clara *et al.* 2010). Phthalates and bisphenol A have been detected repeatedly in several sources that reach surface waters including rainwater, treated and untreated wastewater, sediments, leachates from landfills, and stormwater (Oehlmann *et al.* 2009; Clara *et al.* 2010 and references therein). These chemicals resemble hormones that animals produce naturally (e.g., estrogen) thus they can disrupt a number of reproductive, developmental and physiological processes in animals that ingest or absorb them (Oehlmann *et al.* 2009). In a recent review of available data Oehlmann *et al.* (2009) concluded that both phthalates and bisphenol A have the potential to adversely affect the reproductive biology of wild fish populations. However, few population-level studies have been conducted and population-level responses to such contaminants remain a knowledge gap.

SECTION 7.62: FACTORS AND MECHANISMS THAT CONTRIBUTE TOXIC POLLUTANTS TO AQUATIC ECOSYSTEMS

Surface and groundwater contamination by toxic substances is widespread in developed areas (USGS 1999, 2001; Paul & Meyer 2001; Allan 2004). The diversity of potential contaminants continually increases as new industrial and commercial products and materials are developed and approved for use (Hamilton *et al.* 2004). A thorough review of all known toxic contaminants released within developed areas or their effects on aquatic life is beyond the scope of this report. Instead, IMST provides an overview of toxic inputs, a summary of the diversity of toxic contaminants detected in aquatic ecosystems affected by development, and a description of how contaminant loads accumulate in and distribute throughout aquatic ecosystems. Specific contaminants discussed reflect the availability of published information. Examples of known toxic effects on fishes and food webs and a discussion of the scientific limitations that hinder toxic pollution management are presented in Section 8.1 of this report.

Streams in or Near Developed Areas – In urban and rural-residential areas, numerous point and non-point sources release contaminants into nearby surface and ground waters (Figure 7-2). The following bulleted list provides some specific examples:

- Residential households contribute pesticides, cleaning supplies, pharmaceuticals, personal care products, and other toxic chemicals such as paint, wood preservatives, solvents and various petroleum products (Beasley & Kneale 2002; Ritter *et al.* 2002). Septic systems release chemical contaminants even when these systems are operating properly (Swartz *et al.* 2006; Hinkle *et al.* 2009; ISAB 2007b).
- Impervious surfaces associated with transportation corridors accumulate metals, petroleum compounds, automotive fluids, dioxins, and deicing salts. Stormwater transports these compounds into waterways (Beasley & Kneale 2002; Ritter *et al.* 2002; USEPA 2002b; ISAB 2007b). Airports, even small facilities, apply large quantities of deicing and anti-icing fluids (Corsi *et al.* 2006) releasing glycols and potentially toxic additives into aquatic ecosystems as part of stormwater runoff (Fisher *et al.* 1995; Hartwell *et al.* 1995; Pillard 1995; Cornell *et al.* 2000).

- Outdoor common areas such as golf courses and parks are often maintained using mixtures of herbicides and pesticides (Paul & Meyer 2001).
- Hospitals, extended care facilities, nursing homes, universities, and other civic services contribute a broad array of chemicals associated with research and testing activities, as well as cleaning agents, pharmaceuticals, personal care products, and solvents (e.g., Kummerer 2001).
- Building materials release toxic contaminants as they age. For example, roofing materials are a considerable source of metal contamination (Beasley & Kneale 2002; Ritter *et al.* 2002). Building maintenance often involves application of preservatives that kill degrading organisms (e.g., moss, fungi). Commercial products used for these purposes often contain metals (e.g., zinc) or pesticides that are highly toxic to aquatic organisms.
- Industries contribute a broad array of chemical solvents used in manufacturing and metals are released during waste incineration (Wentz *et al.* 1998; Tanner 2002; Morace 2006). Deposition of particulates from industrial emissions onto impervious surfaces can contaminate stormwater that flows across these surfaces before entering streams and rivers (Beasley & Kneale 2002; Ritter *et al.* 2002). Some industries use large volumes of water for processing and then discharge contaminated effluent to surface waters (Wentz *et al.* 1998).
- Water treatment plants receive contaminants from households, businesses, universities, hospitals, and landfills and route them to surface or ground waters when containment is inadequate or if treatment procedures do not remove toxic substances (Kolpin *et al.* 2002).
- Landfills and hazardous material storage sites (e.g., petroleum products⁵⁵) can release contaminants when containment methods fail (Ritter *et al.* 2002).

Streams outside of Developed Areas – Developed areas can also influence the delivery of toxic contaminants such as fire retardants to surface waters in more remote locations. The number of rural-residential areas in fire-prone ecosystems has been increasing in the western US. These areas have also been associated with increases in wildfire frequency and the overall acreage burned (Sturtevant & Cleland 2007; Syphard *et al.* 2007; White *et al.* 2009). Syphard *et al.* (2007) documented an association between housing density and the size and frequency of human-caused fires in California, with the largest increase in fire frequency being associated with intermediate levels of human disturbance. The effect of fire disturbance and fire retardants on aquatic ecosystems is an active area of research,⁵⁶ but there is little documentation of their cumulative effects on aquatic ecosystems in and around rural-residential developments. Streams flowing through areas burned by wildfires can experience increased water temperatures, reduced dissolved oxygen concentrations, increased loads of naturally occurring toxic compounds (e.g., ammonia) and changes in pH (Crouch *et al.* 2006). Fine fire residues (e.g., ash) can clog the

⁵⁵For examples see ODEQ fact sheets on the Leaking Underground Storage Tanks Program at <http://www.deq.state.or.us/pubs/factsheets.htm#LUST>. Accessed on December 27, 2007.

⁵⁶ See February, 2008 decision on aerial application of fire retardant by the US Forest Service, http://www.fs.fed.us/fire/retardant/Aerial_Application_of_Fire_Retardant.pdf. Accessed on October 10, 2008.

respiratory structures of aquatic organisms and result in negative post-fire effects on aquatic ecosystems (Giménez *et al.* 2004; Crouch *et al.* 2006).

Many retardants used to fight wildfires are ammonium-based formulations that release toxic compounds such as ammonia or cyanide as they break down in the environment (Buhl & Hamilton 2000; Giménez *et al.* 2004). While these compounds break down rapidly and do not bioaccumulate, retardants delivered in or near streams at high concentrations may require significant dilution (e.g., 100-1,750 times for rainbow trout) before they are no longer toxic to aquatic organisms (Gaikowski *et al.* 1996; Buhl & Hamilton 2000; Crouch *et al.* 2006). Limited evidence indicates that accidental delivery of aerially applied fire retardants (e.g., because of erratic winds or poor visibility) to streams and rivers may impair water quality and impose lethal or sublethal effects on aquatic organisms (Buhl & Hamilton 2000; Boulton *et al.* 2003; Giménez *et al.* 2004). Buhl & Hamilton (2000) documented several incidents during the 1990's where the accidental application of fire retardants to streams resulted in fish kills in the John Day River basin (eastern Oregon). Some of these fish kills affected thousands of individual organisms, including hundreds of anadromous steelhead trout (Buhl & Hamilton 2000). Given the potential consequences to aquatic ecosystems posed by fire and fire control strategies, increasing fire frequency in areas undergoing rural-residential development may present risks of unknown magnitude to nearby salmonid populations.

SECTION 7.63: ROUTES OF ENTRY INTO AND RETENTION IN FISH

The chemical properties of toxic pollutants determine how organisms encounter pollutants, how they enter fish bodies, and how they persist and concentrate in fish bodies. Several factors, including the following, make these determinations an exceptionally complex discussion:

- The great diversity of chemical contaminants;
- The considerable variation in solubility of these compounds;
- The wide variation in degradation and transformation rates of different chemicals;
- Differences in geology and water chemistry within and between river systems;
- Differing physiologies and anatomies of fishes;
- Inter-developmental stage differences in fishes;
- Differences in behavior between species and life history stages of fishes;
- Variation in aquatic biota between and within river, stream and estuary systems; and
- Variation of environmental variables through time and between locations, in particular changes in water temperature, pH, sunlight, and hydrologic regimes.

There are two main types of aquatic pollutants, those that are water-soluble (such as salts, minerals, and some metals) and those that are fat (lipid)-soluble (such as PCBs, DDTs⁵⁷, and many other organic pesticides and pharmaceuticals). Solubility properties play a major role in whether individual compounds circulate in the water column or are adsorbed onto sediment

⁵⁷ The abbreviation 'DDTs' denotes the insecticide DDT and its common breakdown products (e.g., DDE).

(Jones *et al.* 2001; Schowanek & Webb 2002). Water-soluble compounds tend to reside in the water column until they degrade. In contrast, lipid-soluble compounds are repelled by water molecules. This chemical behavior increases the rate at which lipid-soluble toxics bind to sediments, slows their environmental degradation (e.g., photolytic or microbial), and increases their persistence in aquatic environments (Ritter *et al.* 2002). Contaminants adsorbed onto microparticles may become distributed throughout the water column which can increase a fish's exposure to the contaminants.

Contaminants generally enter the bodies of fish through their gills during normal gas exchange with the water or with food (Tyler & Jobling 2008). It is also possible for toxic chemicals to be transferred from maternal sources into eggs and into surviving progeny. Once in the bloodstream, toxic effects occur until circulatory, liver, and kidney functions excrete, detoxify, metabolize, or otherwise make the toxic compounds inactive or until the fish dies. Some toxic chemicals (e.g., copper) may only contact a few external cells, but still have severe effects on the behavior or physiology of fish. For example, the receptor neurons that control salmonid olfaction (smell) operate in direct contact with the aquatic environment. Damage to these sensitive cells from contact with copper impairs salmonid sensory perception and affects many ecologically important behaviors such as migration and predator avoidance (Table 7-2).

Both lipid-soluble and water-soluble toxic compounds accumulate in fish bodies. At certain high-energy requiring life stages or during other periods of increased activity, such contaminants or their metabolites could be redistributed as fat reserves are metabolized or other changes in metabolism change the biochemical nature of the fish's body tissues, thereby exposing individuals to elevated risk of toxic effects (Milston *et al.* 2003; Debruyn *et al.* 2004; Svendsen *et al.* 2007). Consequently, some toxics may have little to no biological effect until they are concentrated and then more rapidly released from compartments where they were stored. Certain life stages of fishes are more sensitive to contaminants than others. These tend to be stages of transition, such as during early development or reproduction (Rosenthal & Alderdice 1976). Importantly for salmonids, the smoltification stage (when juveniles migrate to the ocean) can also be very sensitive to toxics (Misumi *et al.* 2005).

Bioaccumulation can occur as prey species that have consumed and retained toxic compounds are subsequently consumed by predators that store toxic compounds at even higher tissue concentrations. Lipid-soluble substances tend to bioaccumulate; whereas, water-soluble contaminants generally do not (Ritter *et al.* 2002). However, some metals (e.g., mercury; Peterson *et al.* 2007) and most organic chemicals have the potential to bioaccumulate. Through this process, contaminant concentrations can increase in a predator's tissues well beyond concentrations measured in surrounding waters or sediments, with the highest concentrations and most severe biological effects found in top predators. The level of exposure depends on how concentrated contaminants are in the water, sediments, and prey, and the volume of each that associates with the predator (e.g., food abundance or gill water contact).

SECTION 7.64: DETERMINING TOXICITY AND SETTING CRITERIA TO PROTECT AQUATIC LIFE

The USEPA sets numeric criteria on chemical concentrations that can be present in surface waters without harming aquatic life. Criteria currently exist for approximately 150 contaminants⁵⁸ but most emerging toxic chemicals lack such criteria. Current information on how toxic contaminants affect aquatic organisms comes, for the most part, from the following four general areas of research and monitoring:

- Acute toxicity lab tests of known contaminants on the time until death of individuals of selected species;
- Chronic toxicity tests of the effects of contaminants on the growth, behavior, and development of test organisms;
- Documenting toxic compounds within the tissues of fish and other aquatic organisms; and
- Correlating toxic chemical loads or documented exposure with negative, sub-lethal effects.

Traditional approaches used to assess contaminant toxicity in aquatic organisms present difficulties for translating individual responses measured in laboratory settings to population-level effects in natural environments. Environmental concentrations of many toxic contaminants are typically below those necessary to complete informative laboratory tests. For example, typical toxicity testing approaches are often insufficient to evaluate the chronic, low-dose exposure typical of many pharmaceuticals and personal care products in aquatic environments (Ellis 2006). This problem occurs because of lack of agreement on appropriate end points to measure, the most sensitive species to test, or the life histories that should be tested, as well as which chemical compounds and complexes to focus on. The need for multiple biological endpoints in toxicity sampling is illustrated by Liney *et al.*'s (2006) research on the fish species *Rutilus rutilus*. Liney *et al.* (2006) exposed early-life stages of *R. rutilus* to prolonged, low doses of municipal effluent and found that genotoxic (affects DNA molecules) and immunotoxic (affects the immune system) effects occurred at much lower doses than doses required to induce changes in the reproductive system.

A Mixture of Contaminants in Aquatic Ecosystems – Developed land uses introduce a diverse array of contaminants to aquatic ecosystems. Kolpin *et al.* (2002) detected multiple pharmaceuticals, hormones, and other organic wastewater contaminants (median = 7 contaminants per sample with a high of 38 contaminants in one water sample) in water samples collected from 139 streams in 30 states. PPCPs, alone, include hundreds of chemicals used for different purposes and with differing chemical activities and environmental persistence. Their ecological effects are largely unknown. The diversity of potential contaminants, particularly PPCPs, increases continually as new products and materials are developed, approved for use (Hamilton *et al.* 2004), manufactured, and purchased by consumers. Method development to detect these compounds in the environment lags far behind the introduction of new contaminants (Jørgensen & Halling-Sørensen 2000). In addition, water quality assessments do not generally

⁵⁸ For detailed information on national water quality criteria for the protection of aquatic organisms visit: <http://www.epa.gov/waterscience/criteria/aqlife/#guide>. Accessed on December 27, 2010.

consider chemical breakdown products or metabolites of parent contaminants. This lack of information makes it difficult to determine actual contaminant loads received by aquatic ecosystems. Pharmaceutical substances are particularly toxic to fish because they are designed to target physiological responses in people. Fish are sensitive to PPCPs because they are vertebrates and vertebrate responses to pharmaceutical compounds provide information to predict physiological effects in fish. However, predicting physiological effects caused by other toxic compounds that were not designed with a targeted vertebrate response can be much more problematic.

Synergistic Toxicities and Additive Effects in the Environment – Streams, rivers, and ground waters contain contaminant mixtures that vary in composition and concentration (Sandahl *et al.* 2005). Mixtures of contaminants are likely to produce synergistic⁵⁹ toxicities (Laetz *et al.* 2009) making it difficult to evaluate the cumulative toxicity of the mixtures. However, little information exists about potential additive⁶⁰, synergistic, or antagonistic⁶¹ effects that may occur in complex contaminant mixtures because most toxicity research focuses on single compounds (Kolpin *et al.* 2002; Sparling & Fellers 2007; Carpenter *et al.* 2008).

The following studies provide information about the cumulative effects of toxicant mixtures. These studies were carried out using contaminant concentrations representative of those commonly experienced by aquatic organisms. Many of the contaminants used in these studies have been found in Oregon (e.g., Wentz *et al.* 1998; Ebbert *et al.* 2000; Fresh *et al.* 2005; ISAB 2007b; LCREP 2007).

- DDT and its breakdown products may exert both endocrine disrupting and immunotoxic effects and may add to the effects of other estrogenic contaminants that alter the reproductive physiology of Pacific Northwest salmonids (Fresh *et al.* 2005 and references cited therein).
- Laetz *et al.* (2009) found that mixtures of organophosphate and carbamate insecticides (e.g., malathion, carbaryl) resulted in additive and synergistic effects on coho salmon nervous system function by inhibiting enzyme activity.
- The herbicide atrazine has been shown to act synergistically with organophosphate insecticides (e.g., diazinon) thereby increasing the toxicities of these substances to aquatic organisms (Anderson & Lydy 2002; Jin-Clark *et al.* 2002; Anderson & Zhu 2004).
- In toxicity tests using an aquatic midge (*Chironomus tentans*), Anderson & Zhu (2004) demonstrated that the toxicity of structurally different organophosphate insecticides increased up to 10-fold in the presence of atrazine.

⁵⁹ The combined effect of two or more contaminants on an organism is greater than the sum of the individual contaminant effects.

⁶⁰ Two or more contaminants in a mixture share a common mode of action and show an effect that is a function of their combined concentration.

⁶¹ The combined effect of two or more contaminants on an organism is less than the sum of the individual contaminant effects.

Table 7-2. Acute and sub-lethal¹ effects of various contaminants on Pacific Northwest salmonids. Contaminants included in this table represent general classes frequently detected in waters affected by rural-residential and urban development and that often exceed criteria established for the protection of aquatic life. Documented effects were typically caused by contaminant concentrations commonly detected in surface waters in the Pacific Northwest.

Ecological Effect	Contaminant Type ²	Common Sources ³	Specific Effects	Biological Level Documented	Selected References
Growth; Development	PCBs	Legacy; Contaminated sediments	Disrupted thyroid function	Individual adult lake trout	Brown <i>et al.</i> 2004
	PAHs	Creosote, asphalt, soot, roofing tar, crude oil, gasoline, and diesel fuel	Reduced biomass, lipid stores	Individual juvenile Chinook salmon	Meador <i>et al.</i> 2006
	PPCPs (e.g., detergents)	Treated wastewater effluent	Reduced growth; Inhibited smoltification; Abnormal reproductive development	Individual Atlantic salmon	Arsenault <i>et al.</i> 2004; Madsen <i>et al.</i> 2004
Sensory; Neurological; Behavioral	Copper	Vehicle exhaust and brake wear; Stormwater transport	Rapid and persistent olfactory impairment; Behavioral processes affected (e.g., migration; predator avoidance)	Individual juvenile coho, chum & Chinook salmon; rainbow trout; Laboratory experiments	Hansen <i>et al.</i> 1999a, b; Baldwin <i>et al.</i> 2003; Sandahl <i>et al.</i> 2004, 2006, 2007; McIntyre <i>et al.</i> 2008
	Organic pesticides: (e.g., insecticides, herbicides, fungicides)	Non-point source transfer from residential and commercial landscaping, right of way maintenance	Reduced neurological enzyme activity; Olfactory impairment; Diverse physiological and behavioral processes affected (e.g., homing; reproductive cues; predator avoidance; swimming speed)	Individual Atlantic, coho & Chinook salmon; rainbow & brown trout	Moore & Waring 1996; Waring & Moore 1997; Scholz <i>et al.</i> 2000, 2006; Jarrard <i>et al.</i> 2004; Sandahl <i>et al.</i> 2004, 2005, 2006, 2007; Jaensson <i>et al.</i> 2007; Tierney <i>et al.</i> 2006a, b, 2007, 2008
Reproductive	Synthetic estrogen (e.g., pharmaceuticals, pesticides and their metabolites)	Treated wastewater effluent	Inhibited testicular growth; Reduced male fertility; Vitellogenin production; Feminization of male fish; Direct negative population level effects	Individual male rainbow trout; fathead minnow	Purdum <i>et al.</i> 1994; Jobling <i>et al.</i> 1996; Schultz <i>et al.</i> 2003; Thorpe <i>et al.</i> 2003; Lahnsteiner <i>et al.</i> 2006; Kidd <i>et al.</i> 2007
	PPCPs (e.g., detergents)	Treated wastewater effluent	Vitellogenin production; Altered hormone production & hormone receptor synthesis; Abnormal reproductive development	Individual Atlantic salmon; rainbow trout; sockeye salmon	Harris <i>et al.</i> 2001; Luo <i>et al.</i> 2005; Bangsgaard <i>et al.</i> 2006

¹Sub-lethal effects are those that alter essential behavior or reduce overall health in ways that can reduce productivity of salmonid populations.

²PCBs – polychlorinated biphenyls; PAHs – polycyclic aromatic hydrocarbons; DDTs – dichloro-diphenyl-trichloroethane and its degradation products (e.g., DDE); PPCPs – pharmaceuticals and personal care products.

³Common sources include those identified in rural-residential and urban environments. Additional, sometimes more significant, sources may also exist outside of developed areas.

Table 7-2 *continued*. Acute and sub-lethal¹ effects of various contaminants on Pacific Northwest salmonids.

Ecological Effect	Contaminant Type ²	Common Sources ³	Specific Effects	Biological Level Documented	Selected References
Immunological	PCBs; PAHs	Legacy; Creosote, asphalt, soot, roofing tar, crude oil, gasoline, and diesel fuel	Immune suppression; Increased disease susceptibility and mortality	Individual juvenile Chinook salmon; Arctic charr	Arkoosh <i>et al.</i> 1998a, b; Maule <i>et al.</i> 2005
	DDTs	Legacy	Long-term immune suppression	Individual juvenile Chinook salmon; Cellular	Milston <i>et al.</i> 2003; Misumi <i>et al.</i> 2005
	PAHs	Creosote, asphalt, soot, roofing tar, crude oil, gasoline, and diesel fuel	Immune suppression	Numerous fish species, including salmonids	Reynaud & Deschaux 2006
Direct Toxicity	Organochlorine pesticides (e.g., Endosulfan)	Insecticide products; Likely transported in stormwater	Nervous system stimulant	Many fish species including salmonids	Gormley & Teather 2003; Wan <i>et al.</i> 2007

¹Sub-lethal effects are those that alter essential behavior or reduce overall health in ways that can reduce productivity of salmonid populations.

²PCBs – polychlorinated biphenyls; PAHs – polycyclic aromatic hydrocarbons; DDTs – dichloro-diphenyl-trichloroethane and its degradation products (e.g., DDE); PPCPs – pharmaceuticals and personal care products.

³Common sources include those identified in rural-residential and urban environments. Additional, sometimes more significant, sources may also exist outside of developed areas.

Toxicity effects have only been determined for a very small fraction of contaminants that may affect aquatic organisms. Laboratory and field-based studies tend to focus on the sensitivity of a few select species to a limited number of contaminants thus ignoring the heterogeneous physiological, predatory, and competitive responses of natural populations. Laboratory toxicity tests also tend to be of short duration, while wild fishes may be exposed to these substances for prolonged periods, often at varying doses, as either discharge of the contaminant into the environment or the volume of recipient waters changes through time (Davis *et al.* 1999; Spromberg & Meador 2006). Consequently, data on the effects of chronic exposure also are lacking for many chemicals and organisms, making ecologically relevant inferences difficult to make (Fent *et al.* 2006).

Setting Criteria to Protect Aquatic Life – The paucity of available information limits the ability to determine protective criteria for aquatic organisms for many of the potentially toxic contaminants and their breakdown products found in waters affected by development (Hamilton *et al.* 2004). Another dilemma for establishing toxicity criteria is that some chemicals may not have biological or environmental threshold effect levels. For example, PPCPs that act as endocrine disruptors and mimic or inhibit action of hormones (Norris 2000; Ellis 2006) may not have a ‘no-effect level’ (Sheehan *et al.* 1999). Recent research on the effects of copper on

salmonid olfactory function also indicates that very low copper concentrations cause cellular damage (Sandahl *et al.* 2007). Also, the biology of fish may not be sufficiently understood to allow recognition of a detrimental effect. For example, understanding the highly relevant process of salmonid smoltification is confounded by the fact that there are no clear anatomical, physiological, or behavioral indicators for when smoltification begins or ends (Stefansson *et al.* 2003).

Even if water quality testing methods kept pace with the production and release of new potentially-toxic contaminants, it is extremely expensive to monitor concentrations and behavior of all the new substances entering developed watersheds. However, the limited information gained from toxicity testing and field monitoring for toxic compounds clearly indicates that contaminants pose serious threats to aquatic organisms including salmonids by affecting reproductive physiology, sensory organs, growth, and development (Table 7-2; Morace 2006). Resolution of these issues is a significant challenge facing efforts to protect water quality in rural-residential and urban areas.

SECTION 7.65: NATURE, TIMING, AND MAGNITUDE OF TOXIC POLLUTION

Urban and rural-residential land uses introduce a diverse array of contaminants to aquatic ecosystems. Not surprisingly, water quality assessments conducted by the USGS NAWQA program (e.g., Hamilton *et al.* 2004; Gilliom *et al.* 2006) and other efforts have documented widespread contamination of streams and groundwater in urban areas. Surveys conducted throughout the Pacific Northwest (Anderson *et al.* 1996; Wentz *et al.* 1998; Ebbert *et al.* 2000; Tanner 2002; Fresh *et al.* 2005; Morace 2006; LCREP 2007; Peterson *et al.* 2007; Carpenter *et al.* 2008) have detected several contaminant types in freshwater and estuarine ecosystems affected by development including:

- Pesticides,
- Fossil fuel byproducts,
- PAHs,
- PBDEs,
- Metals,
- Pharmaceuticals,
- Personal care products, and
- Legacy pollutants (e.g., PCBs; DDTs).

Groundwater surveys also detected various volatile organic compounds (primarily solvents and fuel additives) in up to 80% of shallow monitoring wells beneath urban lands (Wentz *et al.* 1998; Ebbert *et al.* 2000). In Oregon, the cities of Bend, Coos Bay, Eugene-Springfield, La Grande, Medford, and Portland have problems with leaking underground petroleum storage tanks resulting in sites ranked as hazardous by ODEQ (Oregon Progress Board 2000).

Pollutant mixtures that include parent compounds and their breakdown products are common in developed watersheds (Jones *et al.* 2001; Hamilton *et al.* 2004). This is evident in the following results from water quality surveys conducted in the Pacific Northwest and nationwide:

- In a two-year, nationwide survey across 30 states (including Oregon), Kolpin *et al.* (2002) sampled 139 streams susceptible to toxic contamination from human, industrial, or agricultural wastewater and found the following:
 - 86% of pharmaceuticals, hormones and other organic wastewater contaminants tested for were detected at low concentrations;
 - PPCPs were detected in approximately 40% of the 139 streams samples, with 80% of the contaminated streams containing steroids and various non-prescription drugs;
 - Detergent breakdown products, plasticizers, and steroids were found to contribute almost 80% of the total measured concentrations, in over 60% of stream samples; and
 - 75% of streams sampled had more than one organic wastewater contaminant detected (median of seven, maximum of 38).
- For eight urban US streams (including one in Portland, Oregon) surveyed for 75 pesticides, Hoffman *et al.* (2000) determined that 46% of samples contained more than one insecticide currently in use, and 61% of samples contained more than one herbicide currently in use.
- Fifteen percent of urban stream samples collected during NAWQA surveys contained up to 10 different volatile organic compounds and 23% of urban stream samples contained 10 or more pesticides (Hamilton *et al.* 2004).
- Urban streams in King County (Washington) contained 23 of 98 pesticides screened for during water quality surveys (Ebbert *et al.* 2000; Hamilton *et al.* 2004).
- Streams draining mostly urban lands along the Willamette River (Oregon) contained 25 pesticides along with high sediment concentrations of lead, silver, zinc, and cadmium (Wentz *et al.* 1998).

Tissue samples taken from aquatic organisms contain many of the contaminants detected in water and sediment samples, for example:

- In sub-estuaries throughout the Chesapeake Bay (eastern US), PCBs, heavy metals, and other organic contaminant concentrations in fish tissues correlated with the amount of developed land within the watersheds discharging to the estuary (Comeleo *et al.* 1996; King *et al.* 2004).
- In the Puget Sound (Washington), mean concentrations of PAHs, PCBs and fluorescent aromatic compounds were significantly higher in the stomach contents, livers, and bile of juvenile Chinook salmon from an urbanized estuary than in fish from an undeveloped estuary (Arkoosh *et al.* 1998b; Meador *et al.* 2002).
- Mean PCB concentrations in Chinook salmon bred at hatcheries in the Puget Sound (Washington) ranged up to 2.5 times higher than Chinook salmon bred in coastal hatcheries. Fish from the more developed areas of southern Puget Sound had the greatest PCB concentrations (Missildine *et al.* 2005).

- Johnson *et al.* (2007a) measured contaminant concentrations in tissues and stomach contents of coho and Chinook salmon from several Pacific Northwest estuaries and found that Chinook salmon collected from industrial and urbanized estuaries carried the largest contaminant loads. DDTs and PCBs concentrations in the stomach contents and tissues of juvenile Chinook salmon migrating through the Columbia River estuary were among the highest levels recorded to date (Johnson *et al.* 2007a, b; LCREP 2007).
- In Oregon, juvenile salmon sampled from the lower Willamette River and Columbia River estuary contained PBDE concentrations higher than any others measured in the Pacific Northwest (LCREP 2007; Sloan *et al.* 2010).

The following water quality surveys detected contaminant concentrations either exceeding those determined to be protective of aquatic life or experimentally determined to cause harm to aquatic organisms.

- Arkoosh *et al.* (1998b) sampled juvenile Chinook salmon from both a non-urban Puget Sound estuary and an urban estuary where fish were exposed to organic contaminants (PAHs, PCBs) and tested their immunocompetence by exposing them to a common marine pathogen (*Vibrio anguillarum*). Fish from the urban estuary showed higher susceptibility to the pathogen and increased mortality after exposure.
- Nearly all the urban streams sampled during NAWQA surveys contained at least one pesticide in current use at concentrations that exceeded guidelines established to protect aquatic life (Hamilton *et al.* 2004; Gilliom *et al.* 2006).
- Aquatic-life protection criteria were exceeded by the insecticides chlorpyrifos, diazinon, malathion, and parathion in a national comparison of 8 urban streams (including one in Portland, Oregon). Diazinon exceeded water quality protection criteria in 17% of samples (Hoffman *et al.* 2000).
- In Sacramento and Stockton (California), 80% of urban water samples contained chlorpyrifos, and 85% of samples contained diazinon at concentrations exceeding water quality protection criteria set by the California Department of Fish and Game (Bailey *et al.* 2000).
- From 2000 to 2005 in the Clackamas River basin (Oregon), pesticide yields were highest in streams draining urban and industrial lands (compared to agricultural and forested lands), with concentrations of some pesticides (diazinon, chlorpyrifos, azinphos-methyl, DDE) exceeding USEPA and ODEQ criteria for protecting aquatic life (Carpenter *et al.* 2008).
- The following findings were reported for a range of salmonid stocks in the Columbia River estuary:
 - Copper concentrations were detected at levels experimentally shown to inhibit salmonid olfaction (Baldwin *et al.* 2003; Sandahl *et al.* 2007; LCREP 2007).
 - Sediment concentrations of PAHs, PCBs, and DDTs exceeded state or federal sediment quality guidelines. PCBs, PAHs, DDTs, and PBDEs tissue levels in juvenile salmonids approached concentrations determined to be detrimental to their health (Fresh *et al.* 2005; LCREP 2007).

- Pharmaceuticals and personal care products tested for, including some suspected endocrine disruptors, were widespread. Positive tests for vitellogenin⁶² in the tissues of juvenile fish rearing in the Columbia River estuary indicated exposure to endocrine disruptors (LCREP 2007).
- The USEPA (Hayslip *et al.* 2006) measured contaminants (arsenic, cadmium, DDT and its breakdown products, lead, mercury, selenium, and zinc) in the tissues of fish collected from more than 200 randomly selected sites throughout Oregon and Washington estuaries and found the following:
 - In 3.3% of the sampled estuary area, fish tissues contained four chemicals at concentrations exceeding criteria determined to be harmful to fish.
 - In 11.1% of the sampled estuary area, fish tissues contained three chemicals at concentrations exceeding criteria determined to be harmful to fish.
 - In 38.9% of the sampled estuary area, fish tissues contained two chemicals at concentrations exceeding criteria determined to be harmful to fish.

Most of the information available on aquatic pollutants addresses toxic metals and persistent organic contaminants such as PCBs and DDTs. Many contaminants detected in Pacific Northwest waters do not have established aquatic-life protection criteria (Morace 2006) and little is known about the toxicological effects of many of these substances. Other contaminants simply go undetected because they are not screened for during water quality assessments.

Determining the Quantities of Toxic Contaminants that Enter Aquatic Ecosystems – Assessing actual toxic pollutant loads derived from developed landscapes is problematic compared to documenting the diversity of contaminants entering aquatic ecosystems. Pesticides are probably the best understood in this regard. Oregon currently allows the use of 771 active pesticide ingredients in more than 11,000 different products (Carpenter *et al.* 2008). Nonagricultural uses account for ¼ to ⅓ of the total pesticide use in the US (Hoffman *et al.* 2000; Paul & Meyer 2001). Residential areas receive a significant fraction of these compounds (Figure 7-1) and urban application rates tend to be higher (per unit area) than rates documented for agricultural lands (Hoffman *et al.* 2000).

The type and concentration of contaminants found in urban waters may mirror their use and disposal in the surrounding landscape. For example, the active ingredients in insecticide products most frequently purchased in home and garden stores (e.g., the insecticides diazinon⁶³, carbaryl, chlorpyrifos, and malathion) are frequently detected in streams draining urban basins nationwide (USGS 1999; Coupe *et al.* 2000; Ebbert *et al.* 2000; Ritter *et al.* 2002; Hamilton *et al.* 2004; Gilliom *et al.* 2006). The USGS (1999) found that some urban streams contain higher concentrations of these household and garden insecticides than agricultural streams contain farm pesticides. However, limited information exists on the spatial and temporal release patterns of many substances (including pesticides) in urban landscapes and the fraction of these

⁶² Vitellogenin is an egg-yolk protein precursor not normally present at detectable levels in male fish. Tests for this protein are used to assess exposure of male fish to estrogenic contaminants.

⁶³ The USEPA banned residential use of diazinon in 2004;

<http://www.epa.gov/pesticides/factsheets/chemicals/diazinon-factsheet.htm#5>. Accessed on July 18, 2008.

contaminants transported to surface and ground waters (Coupe *et al.* 2000; Hoffman *et al.* 2000; Paul & Meyer 2001).

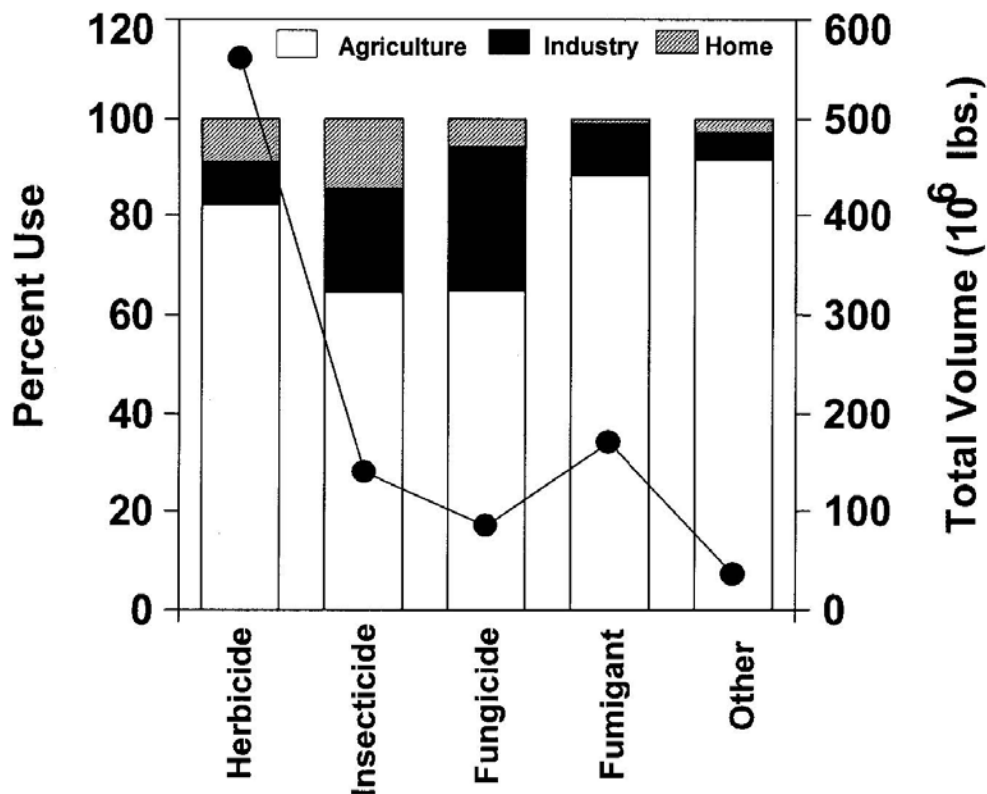


Figure 7-1. Relative use of various pesticides in agriculture, industry, and residential lands in the US. Stacked bars represent percent use and black dots represent total volume. IMST feels that the dots representing total volume should not be connected by the line. Figure reproduced from Ritter *et al.* (2002) with permission from Taylor & Francis Ltd.

Changes in Toxic Contaminants in Developed Landscapes – The spatial and temporal distributions of many contaminants that have been detected in surface waters in the Pacific Northwest are not adequately documented (Wentz *et al.* 1998; Ebbert *et al.* 2000; ISAB 2007b; LCREP 2007). Contaminant concentrations in surface and ground waters depend not only on the quantity released into the surrounding environment, but also on the timing of use, frequency and magnitude of stormwater runoff, and physical and chemical properties of the receiving environment (Hamilton *et al.* 2004). Once released, contaminants move through different environmental paths (e.g., atmosphere, soil, water, wastewater) where they are subjected to microbial, chemical or photolytic processes that modify their concentrations, chemical forms, behaviors, and/or toxicities (Jones *et al.* 2001; Ritter *et al.* 2002; Schowanek & Webb 2002). Contaminants may react with other substances already present and undergo chemical modifications or they may simply break down into alternate chemical structures. Various features in the landscape sequester and concentrate contaminants for variable periods of time.

Detention ponds can sequester heavy metal and hydrocarbon contaminants adsorbed to stormwater sediments (Datry *et al.* 2004; Bäckström *et al.* 2006; Birch *et al.* 2006; Lee *et al.* 2006). Urban estuaries accumulate toxic contaminants from adjacent stormwater runoff and wastewater discharge, as well as from upstream sources (Good 1999; Fresh *et al.* 2005; Lee *et al.* 2006).

The amount of time contaminants persist in the environment depends on their chemical structure and behavior. Some contaminants break down rapidly once they enter surface waters but degradation rates depend on environmental conditions such as temperature or pH. In contrast, metals persist indefinitely and cycle through the environment without breaking down (Ritter *et al.* 2002), although their oxidation states vary depending on aqueous conditions. Some contaminants (e.g., PAHs, dioxins) adsorb onto sediments and break down slowly over time or degrade into breakdown products that can be as toxic as the parent compound. Other contaminants are more water soluble and are broken down more quickly through microbial activity or photolytic degradation. Absorption onto sediment reduces contaminant concentrations in ambient waters but creates an environmentally persistent contaminant source that can degrade water quality whenever sediments are disturbed (Faulkner *et al.* 2000; Beasley & Kneale 2002).

The temporal patterning of contaminant release in watersheds undergoing development is dependent on specific patterns of residential, commercial and industrial land use and the length of time different land uses have been present in a watershed. For example, water quality degradation in the Columbia River estuary likely began with the onset industrialization. Today, the estuary receives contaminants from over 100 point sources in addition to non-point sources from cities like Astoria and Portland in Oregon and Vancouver and Longview in Washington (Fresh *et al.* 2005; LCREP 2007). The contaminant loads contributed by individual sources also change through time. Human usage rates determine the availability of many contaminants such that their release into aquatic ecosystems ranges from episodic to nearly continuous. Wentz *et al.* (1998) found that pesticide concentrations in the Willamette River varied seasonally. Hoffman *et al.* (2000) surveyed eight urban streams and documented carbaryl and diazinon concentrations that frequently exceeded aquatic life criteria during spring and summer months.

The proportional contribution of point sources to pollution loads experienced by aquatic ecosystems depends on the fraction of total flow they contribute to a water way. Ritter *et al.* (2002) reported that the proportions of different point-source pollutants varied considerably throughout the year in 47 Ontario (Canada) urban centers. The fraction of streamflow made up by stormwater was larger during wet weather while the fraction made up by wastewater treatment plant effluents (13 % during wet weather) ranged up to 80% during dry weather (Ritter *et al.* 2002). In a study of water quality of rivers draining to the North Sea, Neal & Robson (2000) found that in rivers affected by urban and industrial development, contaminants with higher water solubility tended to occur in higher concentrations during summer months when low flows reduce the capacity for rivers to dilute pollution. In contrast, contaminants with low solubility or those that bind to sediments exhibit higher concentrations during the winter months when higher flows mobilize sediments (Neal & Robson 2000). An extreme example of this occurred in the Pacific Northwest when a 100-year flood event in 1996 mobilized sediments and associated legacy contaminants in the Columbia and Willamette Rivers. The DDE concentrations in the water column increased to at least 5 times that of the level determined to be protective of aquatic life and in some cases were detected for the first time in some areas (Fresh *et al.* 2005).

Wastewater treatment plants, even those using tertiary treatment processes, do not adequately remove many compounds thought or known to act as endocrine disruptors (e.g., PAHs, personal care products, pharmaceuticals). Consequently, treated effluent acts as a point source for these contaminants (Brooks *et al.* 2006; Conn *et al.* 2006; Belgiorno *et al.* 2007). Kolpin *et al.* (2002) found that concentrations of urban-derived pharmaceuticals were greatest during the low-flow months and undetectable during high-flow months. Brooks *et al.* (2006) estimated that, throughout the US, 23% of treated effluents are released into streams where the effluent dilution factor is less than 10 fold; this fraction increases from 23% to 60% during low flow conditions. In systems where wastewater treatment effluent makes up a dominant portion of flow during part or all of the year, pollutants that pass through treatment processes are a considerable concern. It is technologically feasible for municipal sewage treatment plants to remove pharmaceutical and other toxic compounds, yet this is not done in the US. New technology demonstration projects for this purpose are underway at two municipalities in the United Kingdom (Huo & Hickey 2007). However, the economic costs and benefits of such treatment techniques are unknown.

Over extended periods, clear patterns in the deposition of persistent contaminants can be determined. To document temporal changes of contaminant concentrations in rapidly urbanizing watersheds, the USGS (NAWQA) analyzed sediment layers in urban lakes across the nation (Hamilton *et al.* 2004). Sediment concentrations of tightly regulated or banned contaminants (e.g., lead, DDTs, PCBs) have declined in response to management actions taken to control release of these substances into the environment (Hamilton *et al.* 2004). For example, in a Texas lake lead concentrations peaked in sediments from the late 1960s and declined by about 70% in strata representing periods after the elimination of leaded gasoline. In contrast, nickel concentrations have nearly doubled since 1930 (Beasley & Kneale 2002). Many contaminants, including byproducts from asphalt and fossil fuel combustion (e.g., PAHs), copper, zinc, lead, and chromium have increased in concentration, mirroring increased automobile use in cities and highlighting the importance of impervious surfaces as non-point pollution sources (Mielke *et al.* 2000; Davis *et al.* 2001; Paul & Meyer 2001; Beasley & Kneale 2002; Hamilton *et al.* 2004).

Key Findings: Toxic contaminants

- Surface and groundwater contamination by toxic substances is widespread in and near developed areas. Streams, rivers, and ground waters contain contaminant mixtures that vary in composition and concentration that are likely to produce synergistic toxicities.
- Some toxic contaminants originate from a few significant sources while others originate from numerous low-level sources that make significant contributions when considered in total.
- Substances banned from manufacture and use for years or even decades continue to pollute Oregon's aquatic ecosystems.
- Every year, regulatory agencies approve new compounds for use in the US. Guidelines for general water quality or aquatic life criteria are not determined for most of these compounds prior to their release into surface and ground waters. The paucity of information regarding the occurrence of these contaminants in the environment or their effects on aquatic organisms clouds our understanding of water quality in aquatic ecosystems.
- Traditional approaches to assessing contaminant toxicity in aquatic organisms present difficulties for translating individual responses measured in laboratory settings to population-level effects in natural environments. Little information exists about potential additive, synergistic, or antagonistic effects that may occur in complex contaminant mixtures because most toxicity research focuses on single compounds. Data on the effects of chronic exposure also are lacking for most chemicals and organisms, making ecologically relevant inferences difficult.
- Development of chemical analysis methods for environmental detection of the diversity of potential contaminants that enter aquatic ecosystems or their chemical breakdown products lags far behind the introduction of these compounds, particularly pharmaceuticals and personal care products (PPCPs).
- Most contaminants detected in Pacific Northwest waters do not have established aquatic life protection criteria. Other contaminants simply go undetected because they are not screened during water quality assessments.
- Many water quality surveys conducted in the Pacific Northwest detected contaminant concentrations either exceeding those determined to be protective of aquatic life or experimentally determined to cause harm to aquatic organisms.
- The spatial and temporal distributions of most contaminants that have been detected in surface waters in the Pacific Northwest are not adequately documented.

Section 8.0: Biological Responses of Aquatic Ecosystems to Urban and Rural-residential Development

This section integrates key findings from preceding sections of this report then summarizes what is currently known about how rural-residential and urban developments affect benthic algal and macroinvertebrate assemblages⁶⁴, fish assemblages, and salmonid populations. The previous sections documented how rural-residential and urban developments affect hydrology, physical habitat, water quality, and habitat connectivity (fish passage). These alterations occur simultaneously and exhibit strong interdependency at multiple spatial and temporal scales (King *et al.* 2005; Paul & Meyer 2001; Walsh *et al.* 2005b; Figure 2-1). Collectively these changes impair habitat required by salmonids and other native aquatic biota (e.g., Poff *et al.* 1997; Konrad & Booth 2005) and contribute to the phenomenon referred to as the 'urban stream syndrome' (see below; Meyer *et al.* 2005; Walsh *et al.* 2005b). Several key points from the earlier sections are repeated here for the reader's reference.

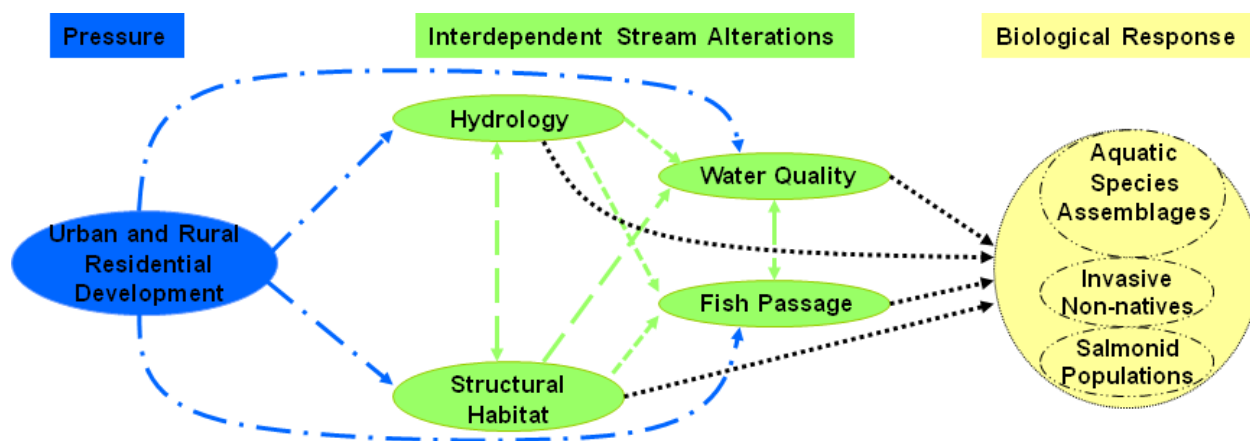


Figure 2-1. The pathways by which urban and rural-residential development alter aquatic communities and the condition of salmonid habitat and populations. Dashed/dotted (blue) arrows indicate direct effects of development on four key components of aquatic ecosystems (green). Dashed (green) arrows depict interactions among components, and dotted (black) arrows represent the influence of the altered ecosystem components on aquatic biota. This figure was first presented on page 39 of this report.

The following are integrated key findings from preceding report sections describing how rural-residential and urban development affects aquatic ecosystems in the Pacific Northwest⁶⁵:

- Rural-residential and urban developments modify processes that link surrounding landscapes to aquatic ecosystems.

⁶⁴ Following Fauth *et al.* (1996), the term assemblage refers to major taxonomic groups within a community (e.g., fish, bird, and insect assemblages). Fauth *et al.* (1996) use the term community to refer to all biota in an ecosystem.

⁶⁵ Because these concepts are discussed in detail earlier in the report, supporting references are not repeated in this summary.

- The volume and chemical composition of stormwater runoff derived from developments have diverse and persistent consequences that can extend throughout entire watersheds and are a major cause of impairment in streams flowing through developed areas. Negative effects of stormwater will increase as human populations and their ecological footprints grow.
- Increasing demands on water resources posed by human population growth and increasing development are significant threats to freshwater ecosystems. Predicted climate changes suggest that water quantity will be reduced and summer stream temperatures will be increased over the next several decades, which could exacerbate these issues in many regions of Oregon.
- Hydrologic responses to development (e.g., increased flashiness, reduced base flow) and changes in physical aquatic habitat (e.g., channelization) can increase channel capacity, stream power, channel erosion, and downstream flooding.
- Vegetation removal from riparian and upland areas and soil disturbance associated with development can increase both the proportion of water that travels through watersheds as surface runoff and the amount of sediment delivered to streams, rivers, and estuaries, while decreasing riparian cover. This combination of actions decreases habitat complexity, impairs water quality, and intensifies the effects of channelization.
- Riparian habitat loss deprives aquatic ecosystems of numerous ecological functions (e.g., buffering water temperatures, reducing erosion, recruiting large instream wood, detoxifying contaminants, sequestering nutrients) that regulate water quality, create physical habitat complexity, and supply nutrients for aquatic food webs.
- Direct modification of in-stream physical habitat (e.g., overwater structures, channel, dredging, wood and aggregate removal) intensifies the effects of altered hydrology and sediment delivery from upland sources.
- Improved wastewater treatment technologies produce progressively cleaner effluent; however, septic systems failures, combined sewer overflows, and large volumes of treated wastewater effluent generated by developments are significant sources of heat energy, organic contaminants, personal care products, pharmaceuticals, and nutrients that enter ground waters, streams, rivers and estuaries.
- For many pollutants, non-point sources contribute much of the pollution entering streams, rivers and estuaries and pose a major threat to water quality.
- The intensity, type, age, distribution of development, and pattern in which individual pollutants move through watersheds introduce variability in the nature and magnitude of water quality degradation.
- Highly engineered stormwater drainage structures and altered channels decrease nutrient retention in streams, wetlands, and riparian areas.
- Physical and physiological barriers impede fish passage within rural-residential and urban areas, but the degree to which these structures limit Oregon's salmonid populations is not well documented.

Section 8.1: *Linking Assemblage and Population-level Changes to Development*

The numerous physical, chemical, and biological factors in aquatic ecosystems that can be altered by development have direct and indirect effects on aquatic organisms. Because these factors are numerous and highly interdependent, it is extremely difficult to isolate individual mechanisms responsible for changes in aquatic species assemblages or populations. At any given time, combinations of various physical and chemical habitat impairments can affect the structure and dynamics of aquatic organism populations. Changes in the population dynamics of individual species subsequently affects the structure of species assemblages in aquatic ecosystems. Aquatic biota responses to development also likely depend on physiographic context (e.g., channel and watershed geomorphology, climate, sediment supply).

Selecting variables useful for characterizing potential assemblage or population-level responses to development is complex for several reasons. First, suites of factors act on assemblages and populations simultaneously. Impervious surfaces, for example, can generate ‘flashy’ pulses of increased flow that not only alter disturbance patterns but also pulse increased toxic contaminant, sediment, heat, and nutrient loads through aquatic ecosystems. Second, the importance of any individual factor in limiting the survival or productivity of aquatic organisms likely varies across time and space. In the Pacific Northwest, the effects of altered hydrology may be stronger during the rainy winter months while water quality impairments (e.g., temperature, dissolved oxygen, toxic contaminants) may be more important during summer low flow periods. Finally, different factors interact in poorly understood ways and possibly produce additive, synergistic, or antagonistic effects on populations or assemblages that are difficult to predict. For species with complex life histories (e.g., multiple life stages with different habitat requirements) responses measured at the individual level or within a single life stage may not translate clearly to population-level changes that are easily measured or monitored (e.g., productivity and abundance). Consequently, simply documenting physical or chemical changes in habitat does not necessarily provide accurate prediction of assemblage or population-level effects.

In the following text box, toxic contaminants are used to highlight the difficulty inherent in linking population- or assemblage-level responses to specific stressors and mechanisms. However, many of the general issues raised also arise when trying to identify the mechanisms that underlie population or assemblage-level responses to changes in hydrology, habitat structure, or other water quality variables associated with development. The effects of development on aquatic ecosystems can be evaluated in experimental streams (Angermeier & Karr 1984; Warren 1971) or by using appropriate survey designs in urban streams (Brown *et al.* 2009; Hughes *et al.* 2004; Roni *et al.* 2008; Pont *et al.* 2009). ODEQ recently published a collaborative study in the Willamette River basin using data from multiple sources and was able to identify major stressors in urban areas such as elevated water temperatures, poor riparian conditions, and unstable streambeds, and related those stressors to macroinvertebrate and fish assemblages (Mulvey *et al.* 2009). However, such studies are expensive, time consuming, and must be repeated in many different areas before broader trends can be accurately characterized. The absence of a clear understanding of such mechanisms means that it is also difficult to predict the effects of alternative development strategies on aquatic organisms in specific locales. The shortage of sufficient data indicates the need for more information on population-level and assemblage-level responses to development in a wide range of ecological settings.

Toxic Contaminants: An example of linking assemblage or population changes to development

Toxic contaminants provide a useful example of the difficulties associated with connecting any particular aspect of development to changes in fish populations or aquatic organism assemblages. Information on the identity of toxic substances, their concentrations in water, sediment, food organisms, or fishes, and the direct or indirect risks they pose to aquatic organisms is only available for a tiny fraction of the thousands of toxic chemicals present in aquatic ecosystems. Depending on their physiology, different aquatic species may exhibit variable responses to the same toxic compounds at different times. Endocrine disrupting chemicals often have non-linear dose-response relationships and affect vertebrate nervous, immune and reproductive systems rather than the typical endpoints of growth or mortality affected by other toxic compounds. The information available on specific responses of individual aquatic organisms to toxic exposure may not extend to all life history stages of the species that have been tested. Demographic characteristics of natural populations may mask or magnify physiological effects that are measurable in individuals (Spromberg & Meador 2006). As a result, high mortality may have little effect on population growth rates for some species, while other species may exhibit population-level effects stronger than what might be predicted from traditional toxicity studies. Tests do exist for traditional toxic compounds, whole effluents, and hormonal activity (Colborn *et al.* 1993; 1996), but the numbers of tests needed make such research an expensive and labor intensive undertaking.

Variable contaminant behavior influences the degree of aquatic organism exposure (Ritter *et al.* 2002). A study of ten common pharmaceuticals including ibuprofen, the anti-epileptic drug carbamazepine, and the anti-parasitic drug ivermectin found that environmental persistence varied widely (Löffler *et al.* 2005). Individual contaminants exhibit variable toxicities depending on environmental characteristics such as water temperature, pH, and salinity (LCREP 2007). For example, trace metal chemistry varies in response to pH, reactions with other inorganic or organic substances, water hardness, carbon dioxide concentrations, and biological processes. Many contaminants move into surface waters along non-point pathways and their presence in surface waters depends on use and climatic and hydrologic conditions in a watershed (Sandahl *et al.* 2005). Contaminant pulses may only last a few hours, and prolonged toxicity depends on the persistence of individual contaminants (Sandahl *et al.* 2005). Toxics may remain stationary if they bind to rapidly settling sediments, or be dispersed far from the point of entry by flowing waters. Mobile organisms can also disperse toxics to locations far from where exposure occurred (e.g., Krümmel *et al.* 2003, 2005; O'Toole *et al.* 2006).

Life-history characteristics of aquatic species determine the extents of their exposure to toxic contaminants (Sandahl *et al.* 2005; Spromberg & Meador 2006). This raises concerns for extending the results of toxicity studies to species with different life histories (Spromberg & Meador 2006). Point sources and sediment settling patterns can create contaminant 'hot spots' that affect aquatic organisms differently depending on the habitat preferences of individual species, populations, or life stages (e.g., Fresh *et al.* 2005). Pacific salmonids vary in their spatial and temporal use of various freshwater and estuarine habitats. The migratory nature of fish, even those normally considered 'resident', confounds our ability to determine when and where individuals may be exposed to contaminants. Migrating fish may experience limited exposure as they move through contaminated areas, while resident fish or fish with longer freshwater or estuarine rearing periods may endure repeated exposure to short-term pulses and long-residence contaminants for weeks, months, or years (Sandahl *et al.* 2005). More sensitive life-history stages (i.e., early developmental, juvenile, and smolt stages; McNabb 1999; Finn 2007) may experience relatively longer exposure periods to various contaminants. For example, out-migrating juvenile Chinook and coho salmon experience widespread exposure to chemical contaminants as they move through Pacific Northwest estuaries, particularly deep draft estuaries with extensive urban development (e.g., Columbia River, Yaquina Bay, Coos Bay) (Morace 2006; Johnson *et al.* 2007a; LCREP 2007). However, Johnson *et al.* (2007a) found that Chinook salmon had whole body contaminant concentrations (PCBs, DDT, PAHs, and organochlorine pesticides) 2 to 5 times higher than corresponding measures from coho salmon. These differences likely reflect variability in residence time, habitat use, and feeding behavior while in the estuary.

Section 8.2: Assemblage-level Responses to Development

In this section, IMST describes the response of assemblages (algal, macroinvertebrate, fish) and populations (salmonid) to anthropogenic changes in urban and rural-residential streams, rivers, and estuaries.

SECTION 8.21: NON-NATIVE SPECIES INTRODUCTION AND ESTABLISHMENT

Developed areas have high rates of non-native species introductions that allow some species to establish self-sustaining populations (Moyle & Light 1996b; Richardson 2000; Witmer & Lewis 2001). A few of the species that become established may become invasive. Invasive species typically reproduce prolifically, disperse widely, and alter native species populations through several mechanisms including competition, predation, and hybridization. These species ultimately modify the structure and function of invaded ecosystems (Lee & Chapman 2001; Tickner *et al.* 2001; Lodge & Schrader-Frechette 2003; Oregon Invasive Species Council 2007). Certain biological and environmental characteristics increase the likelihood that some non-native species will successfully establish and become invasive in new environments. Habitat generalists tend to be more successful as are species that experience little control from diseases, predators, or competitors that would normally limit their reproduction and spread (e.g., Witmer & Lewis 2001; Kennish 2002). Habitat alterations and disturbances may make some sites more vulnerable to invasion and some native species more vulnerable to negative biotic interactions such as competition or predation (e.g., Witmer & Lewis 2001; Meador *et al.* 2003; Zelder & Kercher 2004; Ehrenfeld 2008). Habitat alterations thought to facilitate non-native species invasions include native vegetation removal, changes in water table depth, altered hydrologic flow and fire regimes, soil and water contamination, and changes to stream and river channel morphology. All these alterations can be caused by rural-residential and urban development.

People introduce non-native species into developed landscapes both intentionally and accidentally. Deliberate introductions tend to serve aesthetic, pest control, ecosystem rehabilitation, recreation, or culinary purposes (Table 8-1). State agencies and other entities purposefully stock non-native fish species in reservoirs, lakes, and streams for biological control or recreational fishing. In past decades, many of these non-native fish have naturalized and some have become invasive including common carp (*Cyprinus carpio*), smallmouth bass (*Micropterus dolomieu*), and largemouth bass (*M. salmoides*). Accidental introductions occur when non-native species escape human management or are transported to new locations by ships, planes, recreational boats and trailers, automobiles and trucks, equipment, clothing, luggage, or packaging for live food (Table 8-1). Some non-native organisms are released into new areas when people dump aquaria and non-native pets into natural areas and municipal water systems (Gertzen *et al.* 2008). Non-native disease agents (e.g., whirling disease, *Myxobolus cerebralis*) hitchhike with other introduced species when shipments of non-native plants and animals accidentally include non-native parasites or pathogens (Chapman *et al.* 2003).

At the reach scale, it is difficult to predict where non-native species will establish and spread (Moyle & Light 1996a; Meador *et al.* 2003; Marchetti *et al.* 2006; Ehrenfeld 2008). The incidence and abundance of invasive species does not always increase with increasing human disturbance, because species dispersal is random and unpredictable (stochastic), and because urban land use creates a heterogeneous landscape of microhabitats (Weaver & Garman 1994; Fierke & Kauffman 2006; Moyle & Marchetti 2006; Ehrenfeld 2008; Vidra & Shear 2008). At

Table 8-1. Examples of invasive, non-native species present in Oregon. Species listed were introduced fairly directly by rural-residential, urban, and industrial development¹ and activities related to those areas such as inter-continental commerce. Other species, such as Himalayan blackberry (*Rubus armeniacus*) are associated with developed areas but their introductions are contributed to other land uses such as agriculture and not included here.

Selected Species	Origin; Description	Introduction Pathway into Oregon	Oregon Regions; Habitats	Impact on Aquatic Ecosystems	Selected References
English ivy (<i>Hedera helix</i>)	Europe; ornamental vine	Portland ~1800s; planted into gardens; escaped via birds	Westside; Urban forests	Outcompetes native herbs in forest; Climbs and shades trees	Stoddard <i>et al.</i> 2005; Fierke & Kauffman 2006
Purple loosestrife (<i>Lythrum salicaria</i>)	Eurasia; ornamental herb	North America early 1800s; planted into ponds; escaped via water	Westside; Freshwater habitats	Outcompetes native aquatic plants	Uveges <i>et al.</i> 2002; Zedler & Kercher 2004
Smooth cordgrass (<i>Spartina alterniflora</i>)	E North America; saltmarsh grass	Wash. 1894; accidental import; San Francisco 1970; planted for restoration; Siuslaw ~1970; spread via ballast	Coastal estuaries; Mudflats	Invades eelgrass beds used by salmonid smolts; Changes hydrology of tidal creeks in mud flats	Callaway & Josselyn 1992; Osgood <i>et al.</i> 2003; Nugent <i>et al.</i> 2005; Zedler & Kercher 2004
Green crab (<i>Carcinus maenas</i>)	Europe; estuary crab	San Francisco 1989; accidental import; Oregon ~1997; spread via ballast	Coastal estuaries; Benthic	Feeds on mussels, urchins, barnacles; May alter prey base for salmonid smolts	Nugent <i>et al.</i> 2005; Boersma <i>et al.</i> 2006
Mudsnail (<i>Potamopyrgus antipodarum</i>)	New Zealand; freshwater snail	Snake River 1987, accidental via ships; spread via boats and equipment	Statewide rivers; Benthic	Outcompetes native invertebrates; Covers rocks and other substrates; May alter trophic dynamics that support fish	Nugent <i>et al.</i> 2005
Black bass (<i>Micropterus</i> spp.)	E North America; freshwater fishes	Willamette 1888; stocked for sport; Lake Oswego ~1923; spread via stocking and migration	Statewide rivers; Urban waters	Preys upon juvenile salmonids and other small fish	Hughes & Gammon 1987; Farr & Ward 1993; Gray 2004b; Bonar <i>et al.</i> 2005; Fritts & Pearson 2005
Common carp (<i>Cyprinus carpio</i>)	Asia; freshwater fish	Willamette and Columbia Rivers ~1880; stocked for food; spread via floods and migration	Statewide rivers; Urban waters	Eats roots; Increases turbidity	Gray 2004b; Schade & Bonar 2005; Moyle & Marchetti 2006; Rahel 2007
Mosquitofish (<i>Gambusia affinis</i>)	E North America; freshwater fish	North America ~1900; garden ponds for mosquito control; spread via stocking and floods	Statewide rivers; Urban waters	Eats insect larvae and small fish; Affects trophic dynamics that support fish	Gray 2004b; Schade & Bonar 2005; Rahel 2007
Whirling disease (<i>Myxobolus cerebralis</i>)	Europe; Parasite	Eastern North America ~1950; Oregon 1986; accidental via imported salmonid stocks	Statewide rivers; hatchery fish	infects salmonid species; invades brain causes erratic swimming or death	ODFW 2001; Nugent <i>et al.</i> 2005; Miller & Vincent 2008

¹More complete listings of invasive plants and animals that affect watersheds, riparian areas, or salmonids in Oregon are available through the Oregon Department of Agriculture (www.oregon.gov/ODA/PLANT/WEEDS/), Oregon Invasive Species Council (www.oregon.gov/OISC/), and Oregon Sea Grant (www.seagrant.oregonstate.edu/themes/invasives/). Websites accessed May 24, 2010.

the landscape scale, however, the relationship between development and invasion by non-native species may be more consistent (Riley *et al.* 2005). Leprieur *et al.* (2008) found that the invasiveness of non-native fish species was strongly and positively associated with gross domestic product, percent urban area, and population density. In California, non-native fish distributions were found to be highly and positively correlated with urbanization (Marchetti *et al.* 2004; Riley *et al.* 2005). In Oregon, pollution tolerant non-native fish species were found to be highly abundant in sections of the Willamette River most affected by development (Hughes & Gammon 1987; Wentz *et al.* 1998).

SECTION 8.22: BENTHIC ALGAL ASSEMBLAGES

Algae are photosynthetic organisms that strongly influence dissolved oxygen concentrations and provide a primary food source for aquatic ecosystems. Algal assemblages respond rapidly to changes in water quality and channel substrate. Consequently, bioassessments of aquatic ecosystems affected by development often include some type of algal assemblage characterization (Sonneman *et al.* 2001; Fore & Grafe 2002; Pan *et al.* 2004; Burton *et al.* 2005; Walker & Pan 2006; Weilhoefer & Pan 2006, 2007).

The species composition of algal assemblages can change quickly in response to development (Carpenter & Waite 2000; Pan *et al.* 2004; Walker & Pan 2006; Weilhoefer & Pan 2007). Assemblage changes are highly specific to region and type of disturbance (Fore & Grafe 2002; Potapova *et al.* 2005). The abundance of certain groups of benthic algal species increases as channel structures become simplified and sedimentation is increased. For example, cyanobacteria (or blue-green algae) and green algae can live attached to concrete in modified streams and some benthic diatoms can live in areas of high sedimentation (Carpenter & Waite 2000; Kennen & Ayers 2002; Burton *et al.* 2005). Other algal species respond to changes in water quality that follow development. For example, in the Willamette River basin (western Oregon), the abundance of filamentous green algae increased in wastewater sites with high phosphorus concentrations; cyanobacteria increased in turbid waters containing organic pollution; and diatom species composition shifted in response to increased nutrients in areas of high surface runoff (Carpenter & Waite 2000). In areas where high turbidity, heavy metals, herbicides, or sediment deposition impair water quality, overall algal biomass can decrease even in the presence of high nutrient concentration (Fore & Grafe 2002; ISAB 2007b). Algal species form the basis of many aquatic food webs and changes in algal assemblages have the potential to alter productivity and diversity of species that feed on them. The potential for changes in the trophic structure of aquatic communities in waters affected by development requires additional research.

SECTION 8.23: BENTHIC MACROINVERTEBRATE ASSEMBLAGES

Benthic macroinvertebrate assemblages are rich in species that exhibit diverse responses to changes in aquatic conditions related to various land uses, including urban and rural residential development. Macroinvertebrates provide a major food source for many fishes, including juvenile and resident salmonids, therefore salmonid productivity can be limited by reductions in macroinvertebrate biomass availability (Warren 1971; Hughes & Davis 1986).

Recent studies relating macroinvertebrate assemblage indices⁶⁶ in cool and coldwater streams to urbanization typically report deteriorating index scores as the percent of urbanization increased from 0 to 90% or the percent of impervious surfaces increased from 2 to 60% (Table 8-2) with no clear threshold effects (i.e., there were no levels of urbanization where there was a no-effect level or where assemblage indices began to decline sharply). Results from these studies also show considerable variation in the response indicator (e.g., Index of Biological Integrity (IBI)) at the lower levels of urbanization (e.g., Figure 8-1). Several authors have proposed the following explanations for observed patterns of variability in macroinvertebrate assemblage data:

⁶⁶ Most recent assessments of macroinvertebrate assemblage responses to development employ some sort of index (see Appendix A for detailed descriptions of commonly used indices).

- Single measures of development intensity may not capture all aspects of development that alter aquatic ecosystems (Karr & Chu 1999; Gresens *et al.* 2007).
- Inaccurate estimates of urbanization intensity may result in variability in the response, particularly at low levels of urbanization (Karr & Chu 2000; Tate *et al.* 2005).
- Greater mismatches between the extent of a disturbance and the extent of the response at lower levels of development (Wang *et al.* 2006) or failure to consider landscape ecology variables such as connectivity, proximity, and differing resolutions on predictor and response variables (Steel *et al.* 2010b). For example, Van Sickle *et al.* (2004) found that considering the condition of the entire upstream riparian corridor provided better fit with biological responses than either entire watershed or reach-scale land uses.

Table 8-2. Selected recent literature that reported deteriorating macroinvertebrate assemblage indices with increasing levels of urban development. Literature is restricted to studies published after 1990.

Indicator	Selected References
Index of Biological Integrity (multiple metrics; USEPA 2006c)	Horner <i>et al.</i> 1997; May <i>et al.</i> 1997; Karr & Chu 2000; Booth <i>et al.</i> 2001; Morley & Karr 2002; Stark & Maxted 2004; Cuffney <i>et al.</i> 2005; Limburg <i>et al.</i> 2005; Alberti <i>et al.</i> 2007; Rodriguez <i>et al.</i> 2007
Ephemeroptera, Plecoptera, Trichoptera (pollution intolerant species; Lenat & Penrose 1996)	Richards & Host 1994; Bryce & Hughes 2002; Morse <i>et al.</i> 2003; Ourso & Frenzel 2003; Wang & Kanehl 2003; Gray 2004a; Van Sickle <i>et al.</i> 2004; Carter & Fend 2005; Limburg <i>et al.</i> 2005; Riley <i>et al.</i> 2005; Voelz <i>et al.</i> 2005; Urban <i>et al.</i> 2006; Waite <i>et al.</i> 2008.
Hilsenhoff Biotic Index (pollution tolerant species; Hilsenhoff 1982)	Ourso & Frenzel 2003; Cuffney <i>et al.</i> 2005; Voelz <i>et al.</i> 2005

Others have attempted to improve the performance of predictive variables by relating integrated measures of development to assemblage level responses (McMahon & Cuffney 2000; Brown & Vivas 2005; Tate *et al.* 2005). These estimates integrate distinct types of information about complex development patterns and better account for the numerous aspects of urbanization that may affect aquatic ecosystems (Tate *et al.* 2005). As a result, they hold promise for improving our ability to predict macroinvertebrate assemblage responses to development, particularly at lower development intensities. For example, Alberti *et al.* (2007) reported that both the configuration and amount of development were significantly correlated with macroinvertebrate IBI scores.

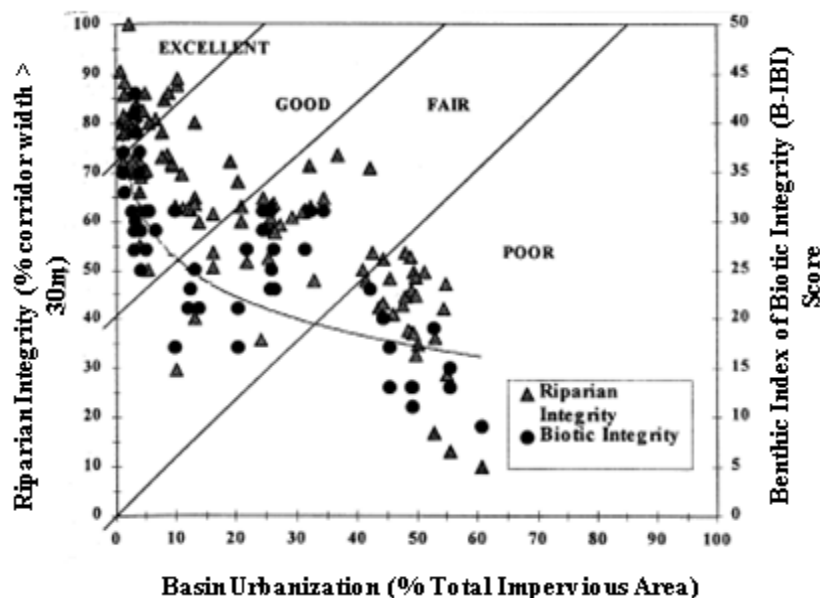


Figure 8-1. Relationship between urbanization and stream biota. Benthic index of biotic integrity (B-IBI) declines with increasing total impervious area, although the correlation is not as strong at lower impervious area. Figure also shows the potential benefits of riparian buffers in urbanizing watersheds. Reproduced from May & Horner (2000) with permission from American Water Resources Association.

SECTION 8.24: FISH ASSEMBLAGES

Fish assemblages respond differently in degree, and to different stressors, than macroinvertebrate assemblages (Yoder & Rankin 1995; USEPA 2000b; Stoddard *et al.* 2005). Therefore, measuring corroborative assemblage changes for macroinvertebrates and fishes can provide complementary evidence of development impacts, and the temporal and spatial scale of those impacts, than either assemblage provides alone (Plafkin *et al.* 1989; Stoddard *et al.* 2005; Davies & Jackson 2006). As for macroinvertebrates, various indices (e.g., IBIs) are commonly used to integrate fish assemblage data (Mebane *et al.* 2003; Hughes *et al.* 2004; Stoddard *et al.* 2005; Davies & Jackson 2006; Pont *et al.* 2006, 2009; Whittier *et al.* 2007a, b). Metrics for coldwater fish assemblages typically include number or percent of salmonids or anadromous fish.

Recent studies relating IBIs or other fish assemblage measures from cool and coldwater streams and rivers to the degree of urbanization consistently showed that the percent of salmonids, the ratio of salmon to trout, and IBI were reduced as the percent of urbanization increased from 0 to 78% or the percent of impervious cover increased from 1 to 78% (Table 8-3). As reported for benthic macroinvertebrates, there were no clear threshold effects found for fish and there was considerable variability found at low levels of development intensity (e.g., Figure 8-2).

Table 8-3. Selected recent literature that reported reduced fish assemblage and population indices with increasing levels of urban development.

Indicator	Selected References
Index of Biological Integrity	Wang <i>et al.</i> 2000, 2001, 2003c; Bryce & Hughes 2002; Drake & Pereira 2002; Mebane <i>et al.</i> 2003; Snyder <i>et al.</i> 2003; Van Sickle <i>et al.</i> 2004; Moerke & Lamberti 2006a; Matzen & Berge 2008; Meador <i>et al.</i> 2008
Percent Anadromous Fish	Limburg & Schmidt 1990, Wolter <i>et al.</i> 2000; Waite <i>et al.</i> 2008
Salmon Spawning and Productivity	Perkins <i>et al.</i> 1980; Scott <i>et al.</i> 1986; Moscrip & Montgomery 1997; Finkenbine <i>et al.</i> 2000; Pess <i>et al.</i> 2002; Regetz 2003; Bilby & Mollot 2008

At the landscape scale, fish assemblages become increasingly homogeneous⁶⁷ as development increases. This pattern is primarily due to the loss of endemic native species, physical habitat impairment, and increases in the abundance and distribution of fish species that are broadly distributed, generalists, or tolerant of poor conditions (e.g., Scott & Helfman 2001; Olden *et al.* 2006; Scott 2006; Light & Marchetti 2007; Smokorowski & Pratt 2007). For example, widely introduced and invasive fish species (e.g., common carp and mosquitofish (*Gambusia affinis*)) contribute to homogenization because their ranges are broad and continue to expand (Rahel 2000, Scott & Helfman 2001, Rahel 2007). Several non-native fish species⁶⁸ have become established and are proportionately more abundant in the Portland reach of the Willamette River (western Oregon) than elsewhere in that river (Farr & Ward 1993; Hughes & Gammon 1987; Hughes *et al.* 2005a, b, c; LaVigne *et al.* 2008). There is also considerable potential for Pacific Northwest fish assemblages to homogenize as urban development increases the occurrence or abundance of non-native aquatic vertebrates, especially if those non-natives are piscivores, such as largemouth or smallmouth bass (Gray 2004b; Bonar *et al.* 2005; Fritts & Pearson 2005; Schade & Bonar 2005).

Non-native fishes can alter native fish assemblages by introducing diseases and parasites, inhibiting native fish reproduction, increasing competition for resources, and increasing predation on native fish (Moyle *et al.* 1986; Baltz & Moyle 1993; Moyle & Light 1996a, b; (Kohler & Courtenay 2002; Marchetti *et al.* 2004). Non-native fishes also can alter habitat, the food web structure, and native fish gene pools (Kohler & Courtenay 2002). In a survey of 31 published studies, Ross (1991) found that native fish populations decreased (and sometimes disappeared) 77% of the time after non-native fish were introduced. Reed & Czech (2005) reported that non-native, invasive species were the second most common cause of fish endangerment in the US. Sanderson *et al.* (2009) estimated that the effects of non-native species on salmonid populations are comparable to those caused by hydropower, hatcheries, commercial harvest, and habitat factors.

⁶⁷ A shift to more uniform species assemblages and communities across the landscape, with a reduction of unique assemblages.

⁶⁸ For example, American shad, (*Alosa sapidissima*), smallmouth bass, largemouth bass, walleye (*Sander vitreus*), yellow perch (*Perca flavescens*), yellow bullhead (*Ameiurus natalis*), and common carp.

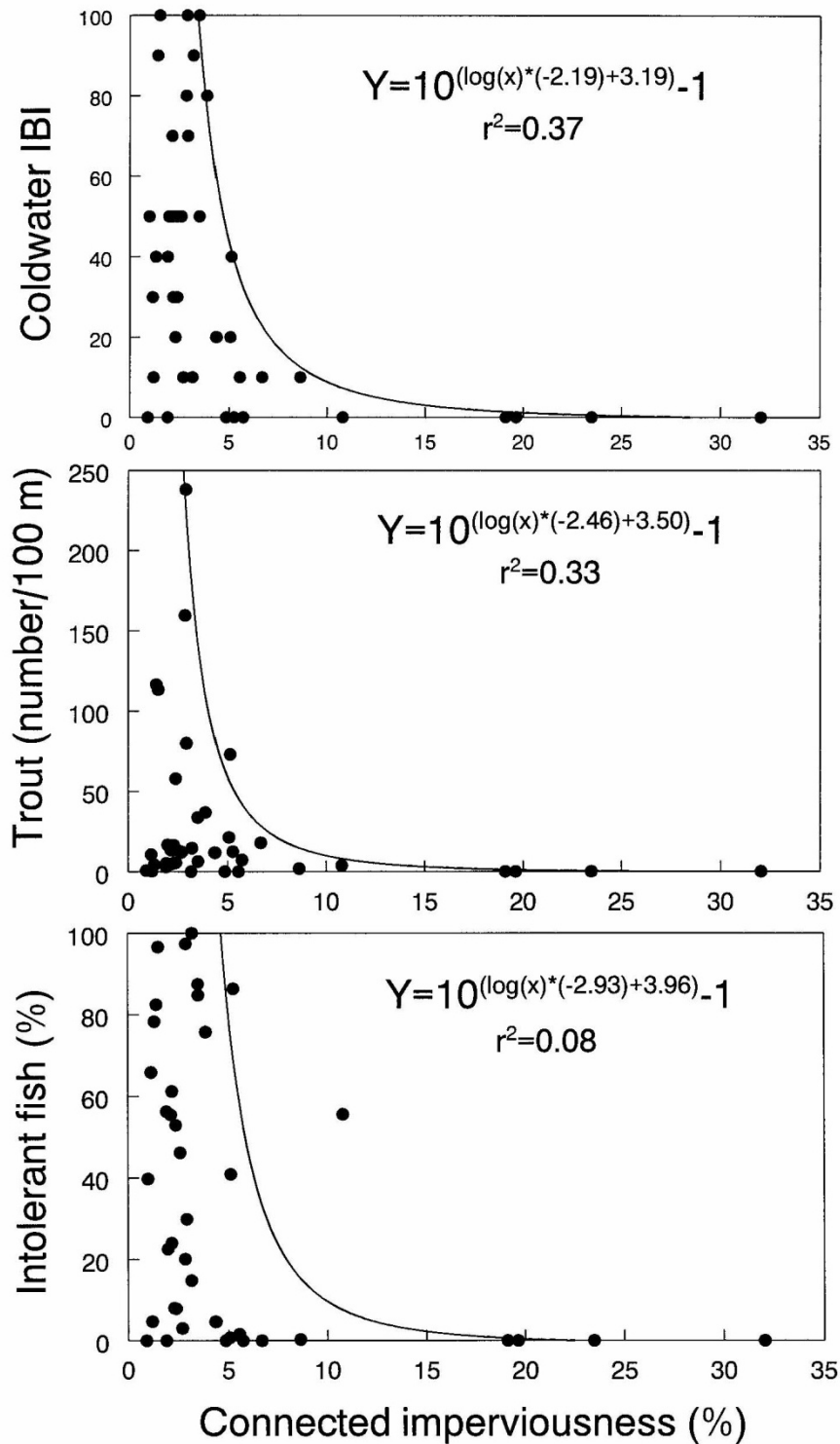


Figure 8-2. Relationships between connected imperviousness (EIA) and the index of biotic integrity (IBI), trout abundance, and percent intolerant fish in Wisconsin and Minnesota coldwater trout streams. Lines and equations on each graph represent 90% quantile regressions fit to the data using a negative exponential model. Figure reproduced from Wang *et al.* (2003b) with permission from the American Fisheries Society.

Because invasive fish are usually found in ecosystems that have already been altered, determining the independent impact of invasive fish on native fish assemblages is difficult (Moyle & Light 1996a; Moyle & Marchetti 2006). Warm-water adapted fish species may become more prevalent in aquatic assemblages as increasing temperatures compromise the reproduction, growth, and survival of common cold and cool-water species (Nelson & Palmer 2007). Temperatures that cause sub-lethal stress are an often overlooked but important component of water quality that can shift species assemblages toward species tolerant of warmer temperatures (Krause *et al.* 2004) and away from salmonids (Reeves *et al.* 1987; LaVigne *et al.* 2008; Sanderson *et al.* 2009).

Section 8.3: Salmonid Population-level Responses to Development

Because salmonids are a focus of this report, this section expands on the effects development can have on salmonids.

Urban and rural-residential developments are frequently located along streams, rivers, and estuaries. Anecdotal evidence, current observations of habitat preferences, historical land surveys, and canning records (Gresh *et al.* 2000) suggest that river reaches flowing through Oregon's urban centers once supported large numbers of salmonids (Groot & Margolis 1991; Stanford *et al.* 1996; Quinn 2005). In the Pacific Northwest, there is a growing understanding that aquatic habitat affected by existing development is important for salmonids (e.g., Pess *et al.* 2002; Regetz 2003; MacCoy & Blew 2005; Sheer & Steel 2006; Burnett *et al.* 2007; Bilby & Molloy 2008). Projections of future land use and land cover in Oregon's coastal mountains show increasing rural-residential and urban development within 328-foot (100-meter) buffers surrounding high quality coho and steelhead habitat, with more rapid development projected for coho habitat (Burnett *et al.* 2007).

Although many studies have characterized habitat preferences or habitat quality for salmonids, most research has associated habitat data collected at or below the stream-reach scale with measurements taken from individual fish. The few studies documenting salmonid population-level responses to development have characterized the presence or health of salmonid populations in relation to land use at the scale of watersheds or river basins. Key findings from these studies include:

- River basins in the Puget Sound (western Washington) region that experienced significant urban development between the 1940/50s and the 1980/90s also experienced salmon abundance declines higher than those in reference basins (Moscrip & Montgomery 1997).
- In the Snohomish River basin (western Washington) adult coho salmon abundance declined on lands converted to urban uses between 1984 and 1998 (Pess *et al.* 2002).
- In a comparison of 22 spring-summer Chinook salmon populations in Oregon, Washington, and Idaho, productivity was lowest in sub-watersheds with more urban land cover (Regetz 2003).
- In the Seattle (Washington) area between 1986 and 2001, coho salmon declined by 75% in index sites affected by development (Bilby & Molloy 2008).

- Also in the Seattle (Washington) area, 24% to 86% of adult coho salmon entering streams affected by urban development die before they spawn compared to approximately 1% in nearby forested streams (Bilby & Molloy 2008). The causes of these high pre-spawning mortality⁶⁹ rates remain under investigation but have been associated with road density (Sandahl *et al.* 2007; ISAB 2007b).
- In a study of 721 Toronto (Ontario, Canada) sites, salmonids declined rapidly as percent impervious cover increased from 0 to 10% and they tended to disappear when percent impervious cover exceeded 10% (Stanfield & Kilgour 2006; Stanfield *et al.* 2006).
- Stranko *et al.* (2008) reported that when areas were only slightly affected by urban development, salmonids disappeared from Maryland streams.
- In the Willamette Valley (western Oregon), the percent of fish assemblages composed of salmonids declined considerably in sites with high levels of urban development (Waite *et al.* 2008).

As discussed in-depth in Sections 4 through 7, development alters hydrology, water quality, physical habitat, and habitat connectivity in ways that not only are detrimental to salmonids and aquatic life in general but are also disproportionate to their land area (Paul & Meyer 2001; Allan 2004; Booth 2005; Brown *et al.* 2005b; Walsh *et al.* 2005b). The loss of spawning, rearing, or refuge habitats equates to reduced species abundance, production, and resiliency in the face of natural and anthropogenic disturbances. Development along lowland streams and estuaries may impose a limiting factor for populations of salmonid species such as chum salmon, coho salmon, and cutthroat trout that rely heavily on lowland habitat for spawning and rearing (Thorpe 1994; May *et al.* 1997; Burnett *et al.* 2007).

Hydrologic alterations and structures intended to stabilize stream banks often convert rivers from complex, multi-channel, meandering paths to simplified, narrow, deep channels that are disconnected from side channels, wetlands, and floodplains (Sedell & Froggatt 1984; Sedell *et al.* 1990; Benner & Sedell 1997). The interaction between active channels and floodplains during lateral channel migration and floods is an important process that creates habitat for salmonids (Amoros *et al.* 1987; Gregory *et al.* 1991). Urban and rural-residential land use actions (e.g., removing riparian and upland vegetation, channelization, bank armoring, inter-basin water transfers, dams, dikes) that alter upland, floodplain, and stream channel hydrology collectively contribute to the loss of habitat for salmonid spawning, rearing, migration, and protection from predators and adverse conditions

Removing riparian vegetation may decrease available habitat created and maintained by large instream wood and lead to increases in water temperatures, sediment inputs, and organic nutrient inputs to streams. Intact riparian areas perform several key functions that create and maintain salmonid habitat including supplying large wood to the stream channel, stream temperature regulation, stream bank stabilization, and regulation of organic nutrient and sediment inputs. Simplification of reach-scale temperature variability results in loss of coldwater refugia that allows organisms to avoid unfavorable temperatures over short periods, thus reducing habitat available for coldwater-adapted salmonids (Poole & Berman 2001). Increased temperature and

⁶⁹ Information available on the NOAA Northwest Fisheries Science Center websites: http://coastalstorms.noaa.gov/stormwater/index.php?option=com_content&task=view&id=8&Itemid=12 and <http://www.nwfsc.noaa.gov/research/divisions/ec/ecotox/movies/cohopsm.cfm>. Accessed on October 12, 2010.

nutrient inputs can cause rapid algal growth resulting in gas supersaturation during the day and substantial oxygen deficits at night, both of which stress salmonids.

Habitat connectivity is affected by fish passage barriers (e.g., dams, diversions, improperly placed and sized culverts, impaired water quality) that restrict or completely block upstream or downstream movement among critical habitats. This can eliminate mobile life history forms from upstream communities (Beechie *et al.* 2006; Neville *et al.* 2006a) and diminish the long-term persistence of populations that become spatially restricted in limited habitats (Wofford *et al.* 2005; Fausch *et al.* 2006).

The degree to which development affects any salmonid population depends on how the species uses habitats within the ecological footprint of a development. In addition to habitat loss, development may limit salmonid population productivity or persistence by:

- Reducing survival,
- Altering behavior, and,
- Altering reproduction, growth, and development.

SECTION 8.31: REDUCED SURVIVAL

Many aquatic ecosystem changes imposed by development affect salmonid survival. Acute lethal events may occur over short periods (e.g., seconds to days) in response to high water temperatures, low dissolved oxygen concentrations, or exposure to high concentrations of toxic substances that interfere with critical physiological functions (e.g., nervous system coordination). In contrast, chronic sub-lethal effects that reduce long-term survival may result from any hydrologic, physical habitat, connectivity, or water quality change that does the following:

- Interferes with normal growth or development;
- Suppresses normal immune function;
- Impairs normal adaptive behaviors (such as predator avoidance, social interactions or migration); or
- Interferes with reproductive development.

Much of the literature documenting salmonid survival in waters affected by development focuses on various water quality impairments. Salmonids require relatively cold water during most life history stages and can experience both lethal and sub-lethal effects from elevated water temperature (Table 7-1; reviewed in McCullough 1999, McCullough *et al.* 2001; IMST 2004). Reduced summer base flows resulting from either municipal water use or decreased groundwater recharge can induce stress-related infections and mortality by increasing water temperature, decreasing dissolved oxygen concentrations, and restricting fish movements to suboptimal locations (Macdonald *et al.* 2000; Rand *et al.* 2006; Newell *et al.* 2007). Elevated suspended sediment concentrations affect both juvenile and adult salmonids but juveniles are particularly vulnerable (Newcombe & MacDonald 1991; Servizi & Martens 1991). Many authors associate acute physiological effects of suspended sediment with gill damage, which causes respiratory impairment and leads to multiple secondary effects (e.g., Newcombe & Flagg 1983; Berg & Northcote 1985; Waters 1995). Suspended sediment effects are mediated by exposure duration

and concentration, with longer exposures and higher concentrations causing greater damage (Newcombe & MacDonald 1991; Servizi & Martens 1992; Newcombe & Jensen 1996). Suspended sediment effects in salmonids include abnormal blood chemistry, weight loss, increased susceptibility to bacterial infection, and death (Newcombe & MacDonald 1991; Waters 1995; Lake & Hinch 1999; Henley *et al.* 2000; Robertson *et al.* 2006).

Toxic contaminants can result in direct mortality or cause a variety of sublethal effects (Table 7-2; Arkoosh *et al.* 1998a). Studies conducted in several Pacific Northwest estuaries affected by development (e.g., McCain *et al.* 1990; Stein *et al.* 1995; Stehr *et al.* 2000) showed that migrating juvenile salmon were exposed to contaminant levels associated with reduced disease resistance and growth (Arkoosh *et al.* 1998a, b, 2001; Johnson *et al.* 2007a, b). Loge *et al.* (2005) estimated that exposure to toxic contaminants in the Columbia River estuary may cause mortality rates of 1.5 to 9% for juvenile Chinook salmon depending on habitat use. Juvenile salmon undergo a period of rapid growth and development and experience many physiological changes during the time they spend in estuaries. Therefore, contaminant exposure may significantly affect the long-term health and survival of these fish as they enter marine environments (Meador *et al.* 2002; Loge *et al.* 2005; Spromberg & Meador 2006; Johnson *et al.* 2007a).

SECTION 8.32: ALTERED BEHAVIOR

Salmonids will actively avoid waters with high suspended sediment concentrations (Bisson & Bilby 1982; Sigler *et al.* 1984; Henley *et al.* 2000; Robertson *et al.* 2007). Cederholm & Reid (1987) found that juvenile coho salmon avoided streams with >4 g/L suspended sediment. High sediment inputs can also bury riffle habitats that are essential for over-winter survival of juvenile salmonids (Bustard & Narver 1975; Hillman *et al.* 1987), and fill pools frequently used by multiple age-classes (Saunders & Smith 1965; Waters 1995), thereby forcing salmonids into sub-optimal habitats. Suspended sediment concentrations also influence foraging behavior and, consequently, growth (Henley *et al.* 2000; Bash *et al.* 2001). While this is generally a negative relationship, with higher suspended sediment concentrations leading to lower growth (Crouse *et al.* 1981; Sigler *et al.* 1984), the dynamic can be complex. Decreased predation risk in turbid waters (e.g., 0.02-0.18 g/L; Robertson *et al.* 2007) may allow juvenile salmonids to spend more time in open water, away from cover (Gregory 1993; Gregory & Levings 1998; De Robertis *et al.* 2003; Newcombe 2003; Korstrom & Birtwell 2006). Increased foraging time may be counteracted by diminished foraging success (i.e., rate of prey capture), leading to net energetic losses and decreased growth. Suttle *et al.* (2004) found that juvenile steelhead foraging success and growth were negatively associated with excess fine sediment because benthic invertebrate prey availability was reduced in highly embedded substrates. In general, streams, rivers and estuaries with elevated suspended sediment loads are less productive than systems with lower suspended sediment loads and are less likely to support large numbers of salmon and trout (Lloyd *et al.* 1987; Waters 1995).

Salmonids possess highly developed sensory systems that play a critical role in predator avoidance, kin recognition, spawning behavior, and migration timing and direction. Some salmonid sensory organs (e.g., olfactory receptor neurons) operate in direct contact with the aquatic environment putting them at risk of damage by water contaminants. Copper and several organophosphate pesticides have been found to impair salmonid olfactory nervous systems (Table 7-2), thus inhibiting their ability to detect chemical signals and respond appropriately.

The concentrations of copper and pesticides used in laboratory experiments generally reflect those detected in waters affected by development (e.g., the Columbia River estuary; Fresh *et al.* 2005). The effects of copper toxicity on the salmonid olfactory system occur rapidly, rendering fish unable to avoid continued copper exposure after the first few minutes (Baldwin *et al.* 2003). Recovery from copper exposure may require several hours to days or weeks depending on the level and length of exposure (Sandahl *et al.* 2007). Because these contaminants are common in stormwater runoff, salmonids can be exposed to repeated contaminant pulses over time.

SECTION 8.33: ALTERED REPRODUCTION, GROWTH, AND DEVELOPMENT

The documented effects of sewage effluents on fish include feminization and likely sterility of male salmonids as well as precocious maturation in females, again likely leading to functional sterility (Kime 1998; Matthiessen 1998; Matthiessen & Sumpter 1998; McNabb *et al.* 1999). Human birth control hormones are estrogenic endocrine disrupting compounds (i.e., they interfere with the normal action of an individual's hormonal system) that are not removed by sewage treatment facilities and are discharged into surface waters with treated wastewater. In terms of toxicity, even low exposure levels to reproductive hormones may have significant implications for salmonid populations (Kolpin *et al.* 2002; Cui *et al.* 2006 and references cited therein; Kidd *et al.* 2007). The breakdown products of detergents (nanophenols) are also mildly estrogenic. Research conducted in Great Britain (Kime 1998) first documented that detergent breakdown products pass through water treatment facilities and act as endocrine disruptors in trout reproduction. Additional experiments have also shown that downstream migration of Atlantic salmon (*Salmo salar*) smolts is delayed when fish are exposed during smoltification (Madsen *et al.* 2004).

Excess sedimentation caused by rural-residential and urban development can significantly reduce the quantity of available spawning habitat in affected streams (Lohse *et al.* 2008). Fine sediments can impair salmonid spawning and incubating habitat by limiting the delivery of dissolved oxygen to developing eggs and alevins, reducing waste product removal, or hindering fry emergence from redds. Galbraith *et al.* (2006) reported that when fine sediment concentrations reached 9 g/L, sockeye and coho salmon egg fertilization rates decreased by $\geq 80\%$. Sediments that settle out of the water column clog interstitial spaces in redds and trap salmonid eggs and alevins in oxygen-deprived environments (Bjornn & Reiser 1991; Waters 1995; Greig *et al.* 2005; Julien & Bergeron 2006; Levasseur *et al.* 2006; Heywood & Walling 2007). High rates of pre-emergence mortality have been observed when fine sediments exceed 15 to 20% of spawning gravel weight (Quinn 2005). Chinook and steelhead egg survival decreased to 5% as fine sediment concentrations increased in a northern California stream (Meyer 2003). The survival and emergence of salmonid fry can be reduced by as much as 50% when fine particles make up more than 30% of spawning substrates (Kondolf 2000).

Altered water temperatures can interfere with adult migration by advancing or delaying it (Quinn & Adams 1996; Cooke *et al.* 2004; Goniea *et al.* 2006) and by creating thermal barriers to upstream movement (Alabaster 1988; Quinn *et al.* 1997). Spawning activity (i.e., redd excavation, egg deposition, and fertilization) may cease if water temperatures are not favorable (McCullough *et al.* 2001; Sauter *et al.* 2001). Temperature governs both development rate and survival of salmonid eggs and alevins (Murray & McPhail 1988; McCullough *et al.* 2001; Nelitz *et al.* 2007). Temperature cues often initiate seaward migrations of juvenile salmonids (Roper &

Scarnecchia 1999; Achord *et al.* 2007). Other temperature-mediated mechanisms that affect salmonid growth include disease resistance, competitive ability, and predation risk (Brett 1956; Poole *et al.* 2001, 2004; Nelitz *et al.* 2007).

There is limited information on the direct effects of altered hydrologic conditions on fishes in urban streams, rivers, and estuaries. The growth and swimming speed of blacknose dace (*Rhinichthys atratulus*) inhabiting urban streams in Baltimore (Maryland) had negative correlations with some aspects of urban development (i.e., impervious surface area) that can alter watershed hydrology (Nelson *et al.* 2003; Nelson *et al.* 2008). If such effects also occur in salmonids, hydrological modifications that follow development could reduce their growth and development.

Key Findings: Biological responses

- It is inherently difficult to link assemblage- and population-level changes to specific (or combinations of) physical, biological, or chemical factors that have been altered by urban and rural-residential development. Few studies have documented salmonid-level responses to urbanization of watersheds.
- Developed areas facilitate intentional and accidental introductions of non-native species. Introduced species may become invasive and can modify the ecological structure and function of native riparian and aquatic ecosystems.
- Benthic algal assemblages can respond quickly to changes in water quality and sedimentation.
- Algal species form the basis of many food webs and changes in assemblages have the potential to affect the trophic structure of aquatic ecosystems.
- Macroinvertebrates provide a major food source for many fishes, including juvenile and resident salmonids; therefore salmonid productivity can be limited by reductions in macroinvertebrate biomass availability. Benthic macroinvertebrate assemblages are rich in species that exhibit diverse responses to changes in aquatic conditions related to various land uses, including urban and rural residential development.
- At the landscape scale, fish assemblages become increasingly homogeneous as development increases because of the loss of endemic species, aquatic habitat impairment, and an increase in non-native fish.
- Development may limit salmonid population productivity or persistence by reducing habitat quality, quantity, and accessibility; by reducing reproduction rates, survival, growth, and development at various life-stages; and by altering behavior such as foraging and migration.