TECHNICAL MEMORANDUM

Date: September 16, 2022

To: Minnie Dhaliwal, Director

From: Greg Johnston, Senior Fisheries Biologist
Clover McIngalls, Environmental Planner
Dan Nickel, Environmental Engineer

Project Name: Issaquah Streams BAS Review- Phase I

Project Number: 220629

Subject: Issaquah Critical Areas Ordinance (CAO) Stream Buffer Widths- Best Available Science Summary and Regulatory Considerations

Introduction

GMA Regulatory Update Process

The City of Issaquah is conducting a substantive review and revision of its Critical Areas Ordinance (Land Use Code Title 18, Chapter 10), as part of a city-wide Land Use Code update effort aimed at improving the regulations that guide growth and development in Issaquah. The Growth Management Act (GMA) under RCW 36.70A.172 requires all counties and cities in Washington State to "include the best available science in developing policies and development regulation to protect the functions and values of critical areas." The information contained in this memorandum is a summary of the best available science (BAS) source documents related to stream riparian area protection, applicable to the update of stream buffer requirements in the streams section of the CAO.

Purpose and Overview of this Memo

This memo presents a summary of existing conditions of streams and riparian areas in the City of Issaquah as documented in past studies, and the BAS related to stream and riparian area functions and protection. Past studies reviewed and summarized include an (Issaquah) Stream Inventory and Habitat Evaluation Report (Parametrix. 2003), an (Issaquah) Stream and Riparian Areas Restoration Plan (Watershed Company, 2006), and a City of Issaquah Best Available Science Report (Author, 2004).

The proposed Title 18 code updates related to streams are discussed and appropriate riparian area buffer widths are recommended that are aligned with the referenced BAS.

Existing Conditions Summary

The City of Issaquah is located within the Cedar-Sammamish watershed in Watershed Resource Inventory Area (WRIA) 8. This watershed drains 692 square miles in King and southern Snohomish counties and contains two major river systems (The Cedar River and The Sammamish River) and three large lakes (Lake Washington, Lake Sammamish, and Lake Union) (WRIA 8 Salmon Recovery Council 2017). The Lake Sammamish subbasin, in which the City is located, includes Issaquah, Tibbetts, Laughing Jacobs and Lewis Creeks, as well as many smaller tributaries. The following sections are comprised primarily of the basin descriptions provided in the City's past studies (Watershed 2006, Parametrix 2003) that were reviewed for this memo. Descriptions have been supplemented where new or changed information is known.

Issaquah Creek and Tributaries

The Issaquah Creek basin is located within the Lake Sammamish Watershed (as are all of the stream basins partially or wholly within the City) and encompasses approximately 61 square miles, including the North Fork and East Fork of Issaquah Creek and Tibbett's Creek (Figure 1). Tibbett's Creek was included in the Issaquah Creek basin by King County even though it flows directly to Lake Sammamish. The mainstem of Issaquah Creek originates in the foothills of the Cascades on the slopes of Tiger Mountain and flows northerly for approximately 17 miles to the mouth. Most of the basin is forested, with the remainder being developed or occupied by pastures, fields, and/or wetlands. The mixed deciduous/coniferous forests, streams, wetlands, and fields within the basin provide significant habitat values for a wide range of aquatic and terrestrial species. The shoreline of mainstem Issaquah Creek within the city limits is dominated by urban uses, as described below.

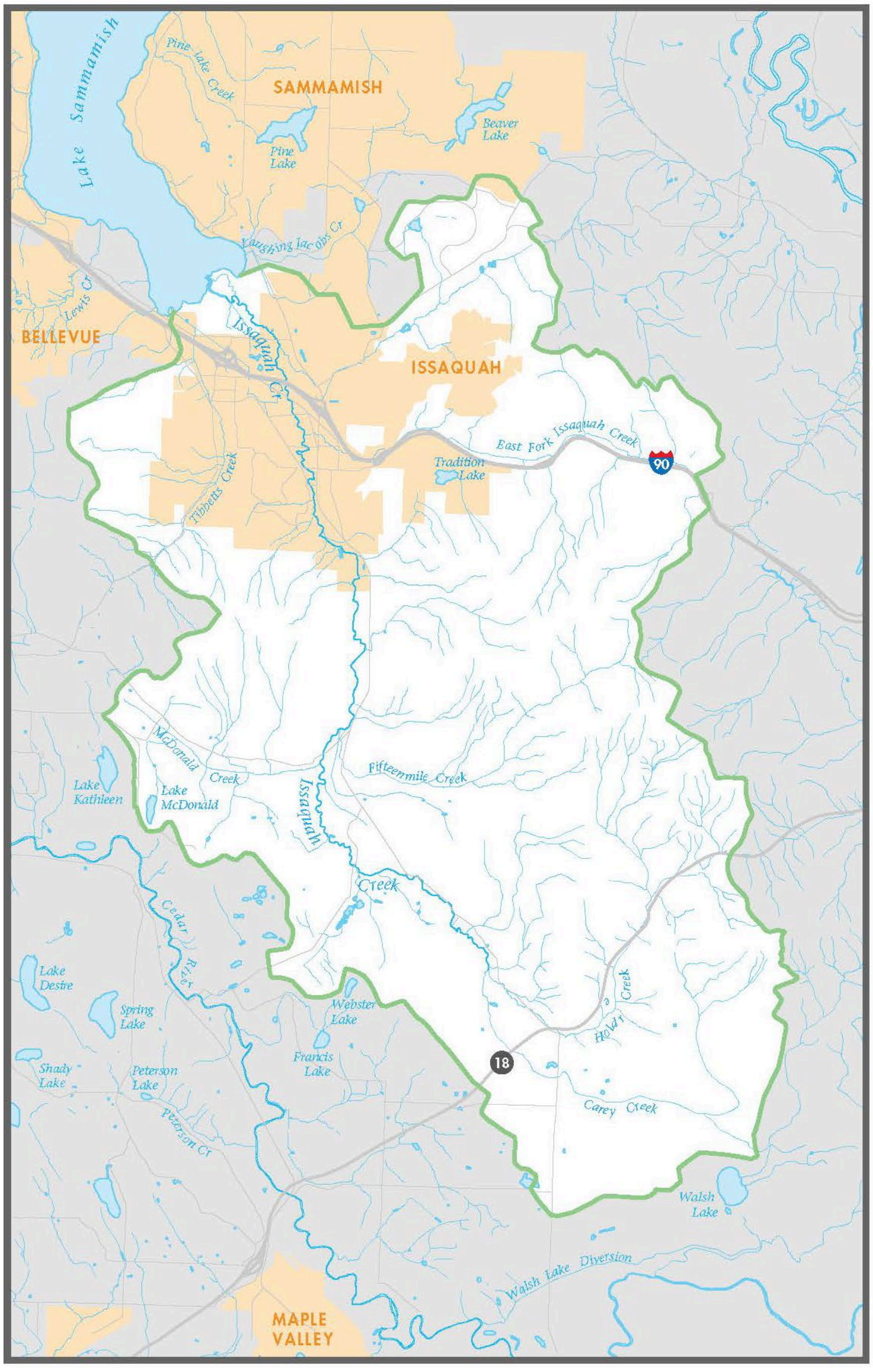

Figure 1. Issaquah Creek basin, including Tibbetts Creek. King County, accessed Aug. 2022.

Issaquah Creek and its tributaries are used by a number of salmonid fish species including chinook, coho, and sockeye salmon (including kokanee); cutthroat and steelhead trout; and occasionally bull trout. Large numbers of adult coho and Chinook salmon return each year to the Washington Department of Fish and Wildlife (WDFW) Issaquah Salmon Hatchery which is located along Issaquah Creek at River Mile 3.1, in the heart of the City. Many salmon are also allowed to pass upstream beyond the hatchery into the upper basin to spawn, making use of the available spawning and rearing habitat. Adult upstream migrants need places to rest and hide from predators, as do juveniles. Some adult fish inevitably stop short of reaching the hatchery to spawn, so suitable spawning habitat below the hatchery is in high demand. Downstream juvenile migrants, as well as some juveniles who rear for longer periods within the stream, also need functional rearing habitat.

Land use along the shoreline of mainstem Issaquah Creek is a mix of commercial, residential, and community facilities. The shoreline in the downstream regions, north of SE 56th St., consists primarily of Lake Sammamish State Park, shifting to mainly commercial properties, and then increasing residential properties moving upstream beyond (south of) NW Gilman Bvd. Impervious surfaces in the creek corridor reflect zoning and land use: the areas near the downtown commercial center have a higher proportion of impervious surface area than areas with less intensive land use. Multiple, large, restoration projects are currently planned or are on-going including in-stream habitat and natural process improvements along the approximately 6,600-foot stretch of lower Issaquah Creek that flows through Lake Sammamish State Park, sponsored by Mountains to Sound Greenway Trust and planned for implementation in 2023/24. The City of Issaquah is currently (September 2022) in construction on an in-stream and riparian habitat improvement project along approximately 1,200 feet of Issaquah Creek just north of I-90. Both projects are designed to restore fish and wildlife habitat in Issaquah Creek and its riparian areas by strategic placement of large woody debris, improving floodplain connectivity, and riparian planting.

Very high-gradient valley wall streams deliver sediment to the upper reaches of mainstem Issaquah Creek and its headwater streams. Increased runoff and loss of channel structure in some of these steep side tributaries has increased sediment supply to the delta fans at their mouths. Increased runoff, loss of channel bank structure, and channel confinement provide more energy to move the ample sediment supply downstream into mainstem Issaquah Creek, which is primarily sandy substrate and sand overbank deposits (Parametrix 2003).

Laughing Jacobs Creek

Laughing Jacobs creek headwaters on the Sammamish Plateau within the City of Sammamish before descending into the City of Issaquah through a steep ravine. Within the ravine, near the City limits, is Laughing Jacobs Falls, which marks the upstream extent of anadromous fish use. The stream continues downslope and crosses East Lake Sammamish Parkway, with its mouth lying just north of the boat launch at Lake Sammamish State Park. Its entire basin encompasses approximately 3,600 acres (over five square miles), and includes two lakes, Beaver Lake and Laughing Jacobs Lake. The upper basin, in the City of Sammamish, is mostly sloped terrain with mixed residential uses interspersed with forested areas. In Issaquah, the creek flows through mostly-forested ravine areas, the Hans Jensen Group Camp area at Lake Sammamish State Park, and then crosses East Lake Sammamish Parkway and several private properties prior to entering Lake Sammamish. Laughing Jacobs Creek is used by kokanee and coho salmon along its lower reaches and cutthroat trout throughout. Habitats at and near the mouth are also known to be used by juvenile Chinook salmon.

Tibbetts Creek

Though it enters Lake Sammamish separately, Tibbetts Creek is referred to as a sub-basin of Issaquah Creek in the *Issaquah Creek Current/Future Conditions and Source Identification Report* (King County Surface Water Management Division 1991). The Tibbetts Creek basin encompasses approximately five and a half square miles and borders the west side of the Issaquah Creek basin proper. Tibbetts Creek is slightly more than four miles long and originates on Squak Mountain at an elevation of 1,080 feet. Headwater sections are steep and confined, flowing through ravines, with a broader floodplain developing about mid-way along its path to Lake Sammamish. The upper portion of the stream and its tributaries are steeply graded with numerous cascades. Much of the lower portion, which is located on the broad floodplain, is low-gradient and channelized as it flows through Issaquah.

Vegetation along the Tibbetts Creek corridor includes greater percentages of western red cedar and Douglas fir compared with the riparian areas of lower mainstem Issaquah Creek and the East Fork Issaquah Creek, which are in more developed areas. Riparian upland vegetation communities, particularly in the northwest portions of the City, are dominated by salmonberry and sword fern.

Efforts to restore Tibbetts Creek have been underway since the late 1990s. Mine tailings removal, relocation and restoration of 1,800 feet of stream channel, and numerous other in-stream and riparian habitat enhancement projects have been completed by both public and private interests.

Tibbetts Creek is used by the same salmonid fish species as Issaquah Creek and its tributaries, though Chinook and steelhead use is only occasional, and bull trout use is infrequent throughout all of the City's streams. A variety of non-salmonid fish species also use these creeks.

Zoning and land use along the Tibbetts Creek corridor are mostly low-density residential but also include City property, retail, and professional office space.

Development and modifications to the physical channel have contributed to the deterioration of Tibbetts Creek instream habitat. Lower Tibbetts Creek is depositional, and natural channel migration occurs as sediment transported from higher in the subbasin is deposited on the valley floor. However, the loss of channel mobility coupled with significant increases in sediment load has resulted in substrate dominated by silt and sand, which is not good salmonid habitat. Channelization has also resulted in the loss of habitat complexity, particularly rearing habitat, and lowered overall salmonid productivity (King County SWM 1996).

Lewis Creek

The Lewis Creek basin encompasses approximately two square miles on the north side of Cougar Mountain, flowing off the mountain towards Lake Sammamish in a steep, narrow ravine. The upper portions of the basin lie primarily within the City of Bellevue, with a few areas still remaining in unincorporated King County. The lower basin, including all of the areas downstream of I-90, lies within the City of Issaquah. The connected culverts under I-90 and adjacent to West Lake Sammamish Parkway to the north and Newport Way to the south are approximately 1,000 feet in combined length and constitute a complete barrier to upstream fish passage.

Between the base of Cougar Mountain at I-90 and Lake Sammamish, the stream gradient decreases to approximately four percent as the creek flows for approximately 0.6 miles across a broad fan of seismically uplifted lake bottom sediments (King County DNR 2006). This area is occupied by moderate-density residential development. Lewis Creek is used by kokanee and coho salmon, and cutthroat and steelhead trout. Of these, kokanee are the fish species of primary concern due to their decline throughout the Lake Washington basin in general and Lake Sammamish drainages in particular. Unlike many streams in these watersheds, however, Lewis Creek still maintains a relatively strong, though variable, run of these fish and so is important for the overall preservation of Lake Sammamish kokanee.

Best Available Science Summary

Functions and Potential Effects of Development

WDFW defines riparian ecosystems as, "…transitional between terrestrial and aquatic ecosystems and are distinguished by gradients in biophysical conditions, ecological processes, and biota. They are areas through which surface and subsurface hydrology connect waterbodies with their adjacent uplands. They include those portions of terrestrial ecosystem that significantly influence exchanges of energy and matter with aquatic ecosystems" (Quinn et al. 2020). Specific stream and riparian area functions are discussed below.

Recruitment of large woody debris to the stream

Large woody debris (LWD) plays a significant role in geomorphic functions such as directing stream flows to shape the channel form and influencing sediment storage, transport, and deposition rates. The many effects of large wood create a variety of channel morphologies—dam pools, plunge pools, riffles, glides, undercut banks, and side channels - which provide a diversity of aquatic habitats (Quinn et al. 2020). Instream large wood provides fish with cover from predators, and by increasing a water body's effective space, wood structures may increase fish densities (Bisson et al. 1987). Simply put, large wood provides the downward scour necessary for streams to create pools, and it provides protective cover for fish in those pools. Pools provide rearing habitat for juvenile fish and resting space for adults.

The collection of large woody debris and the subsequent entrapment of smaller branches, limbs, leaves and other material reduce flow conveyance in small streams and increase temporary flood storage (Dudley et al. 1998). By retaining smaller organic debris, LWD provides substrate for microbes and algae, and prey resources for macroinvertebrates (Bolton and Shellberg 2001). Just as riparian areas have a more significant effect on smaller channels compared to larger channels (Vannote et al. 1980), the effects of LWD in small channels are particularly significant (Harmon et al. 1986). In small channels, LWD provides important structures in the stream, controlling rather than responding to hydrologic and sediment transport processes (Gurnell et al. 2002). For this reason, large wood is responsible for significant sediment storage in small channels (Nakamura and Swanson 1993, May and Gresswell 2003), thereby increasing channel stability (Quinn et al. 2020). The significant role of large wood for storing sediment is further revealed where wood has been experimentally removed from streams and sediment is mobilized and storage reduced (Bilby 1981). Large wood that partially blocks flow can also help to encourage hyporheic flow through the streambed substrate (Poole and Berman 2001, Wondzell et al. 2009).

Instream large wood provides for a wider range of flow velocities, in turn resulting in a diversity of aquatic habitats - pool formation, streambed scour, sediment deposition, and channel migration (Quinn et al. 2020). As such, large woody debris plays an important role in forming complex in-water habitat structures that provide flow refugia and essential cover and improved foraging conditions for fish. Fausch and Northcote (1992) found that streams containing large amounts of LWD supported populations of juvenile cutthroat trout and coho salmon five times greater than streams within the same river system that had been cleared of LWD. Roni and Quinn (2001) found that winter densities of coho salmon, steelhead, and cutthroat trout were higher in streams where LWD had been added.

Large wood recruitment is often the result of bank erosion, windthrow, landslides, debris flows, snow avalanches, and tree mortality due to fire, ice storms, insects, and disease (Swanson et al. 1976; Maser et al. 1988). Large woody debris can enter channels through individual trees falling into the stream, as well as through larger disturbances (Bragg 2000). A comparison of 51 streams with varying channel form in mature forests of British Columbia found that of the approximately one-third of LWD pieces for which the source could be identified, tree mortality was the most common (65 percent) entry mechanism (Johnston 2011). Streambank erosion and associated channel migration is a common method of wood recruitment in large alluvial channels (Murphy and Koski 1989), whereas in smaller, steeper channels, wood recruitment predominantly occurs through slope instability and windthrow (May and Gresswell 2003).

The probability of a tree entering the channel decreases with distance from the streambank (McDade et al. 1990, Grizzel et al. 2000). Past research has found that most LWD originates within approximately 30 m (98 ft) of a watercourse (Murphy and Koski 1989, McDade et al. 1990, Van Sickle and Gregory 1990, Robison and Beschta 1990). In 90 percent of the 51 streams surveyed in British Columbia, 90 percent of the LWD at a site originated within 18 m (59 ft) of the channel (Johnston 2011). May and Gresswell (2003) found that wood was recruited from distances farther from the stream channel in small, steep channels (80 percent from 50 m (164 ft) from the channel), compared to broad alluvial channels (80 percent from 30 m (98 ft) from the channel) because of the significance of hillslope recruitment in narrow valleys.

The likelihood of downstream transport of LWD is dependent on the length of wood relative to bankfull width of the stream (Lienkaemper and Swanson 1987). Wood that is shorter than the average bankfull width is transported more readily downstream compared to wood that is longer than the bankfull width (Lienkaemper and Swanson 1987). Therefore, large wood is rarely transported downstream from small channels less than 5 m (16 ft) in width (May and Gresswell).

Beaver dams incorporate both small and large wood, and serve to slow water, retain sediment, and create pools and off channel ponds used by rearing coho salmon and cutthroat trout (Naiman et al. 1988, Pollock et al. 2004). The removal of these structures throughout history has been linked to a significant reduction in coho salmon summer and winter rearing habitat in the nearby Stillaguamish River (Pollock et al. 2004).

Shade and microclimate

Stream temperatures and riparian microclimate conditions are closely tied to each other. Factors influencing water temperature and microclimate include shade, orientation, relative humidity, ambient air temperature, wind, channel dimensions, groundwater, and overhead cover.

Salmon and other native freshwater fish require cool waters (55-68°F) for migrating, rearing, spawning, incubation, and emergence (USEPA 2003). Thermal tolerances differ by species; coho salmon prefer the coolest temperatures, whereas steelhead can tolerate higher temperatures. Riparian microclimate affects many ecological processes and functions, including plant growth, decomposition, nutrient cycling, succession, productivity, migration and dispersal of flying insects, soil microbe activity, and fish and amphibian habitat (Brosofske et al. 1997). Amphibians have narrow thermal tolerances, and they are particularly influenced by changes in microclimate conditions (Bury 2008).

Several studies have documented significant increases in maximum stream temperatures associated with the removal of riparian vegetation (Beschta et al. 1987; Murray et al. 2000, Moore et al. 2005, Gomi et al. 2006).

A number of studies have considered the extent to which different riparian zone widths modulate stream temperature. In headwater streams in British Columbia, 10 m (33 ft) riparian zones generally minimized effects to stream temperature from timber harvest, although maximum daily temperatures reached 3.6°F higher than control streams (Gomi et al. 2006). A comparative study of 40 small streams in the Olympic Peninsula found that mean daily maximum temperatures were 2.4°C higher in logged compared to unlogged watersheds, and that logged watersheds had greater diurnal fluctuations in water temperatures (Pollock et al. 2009). Another study of streams in Washington found that stream temperatures were most closely correlated with vegetation parameters associated with the riparian area, such as total leaf area and tree height, and that the effect of buffer width was less significant, particularly for buffers larger than 30 m (98 ft) (Sridhar et al. 2004). These findings are consistent with an earlier study relating angular canopy density, a proxy for shading, to riparian buffer width; which found that the correlation between shade and riparian buffer width increases up to around 30 m (98 ft) (Bestcha et al. 1987). Therefore, for buffers less than 30 m (98 ft), buffer width is expected to be more closely related to shading and stream temperatures than buffers over 30 m (98 ft).

Riparian buffers necessary to maintain forest microclimate are controlled by edge effects, which tend to extend well into forested areas adjacent to clearings. However, riparian buffers ranging from 10-45 meters in width may minimize microclimate effects related to light, soil, and air temperatures. A study of small streams in Western Washington indicated that buffers greater than 45 m (147 ft) wide are generally sufficient to protect riparian microclimate in streams (Brosofske et al. 1997).

Bank integrity

Riparian vegetation helps provide bank stabilization through a complex of tree roots, brush, and soil/rock. Woody vegetation tends to provide greater bank stability than herbaceous vegetation because woody vegetation has larger roots that extend deeper into the streambank (Wynn and Mostaghimi 2006).

Bank stabilization functions are at a higher risk of degradation in urbanized watersheds. As with sediment reduction, the streambank stabilization functions of vegetation increase with buffer width out to approximately 80 to 100 feet; after this point, disproportionately large increases are needed to improve riparian function (Castelle and Johnson 1998).

Runoff filtration and nutrient inputs (water quality)

Water quality is characterized by several physical, chemical, and biological factors, including suspended sediment, nutrients, metals, pathogens, and other pollutants. Water quality characteristics are controlled by upslope, as well as riparian conditions. This section discusses how water quality is maintained under natural conditions. Water temperature is also a component of water quality, which is addressed separately.

When development results in reduced infiltration and increased surface flows, sediment and contaminants are transported more directly to receiving bodies without interfacing with natural soil filtration processes. Because of this, urban areas tend to contribute a disproportionate amount of sediment and contaminants to receiving waters relative to the percentage of urbanized area within the watershed (Sorrano et al. 1996). Heavy metals, bacterial pathogens, as well as PCBs, hydrocarbons, and endocrine-disrupting chemicals are aquatic contaminants that are commonly associated with urban and agricultural land uses.

The full suite of sublethal and indirect effects of these contaminants and combinations of contaminants on aquatic organisms is not fully understood (Fleeger et al. 2003). Likely some contaminants with potentially severe repercussions for fish and wildlife have yet to be identified. For example, research in the Puget Sound region had identified mature coho salmon that return to urban creeks and die prior to spawning, a condition called pre-spawn mortality (Feist et al. 2011, Sholz et al. 2011). After a prolonged and diligent investigation, the specific cause of the condition has been recently attributed to 6PPD-quinone, not specifically a component of tire formulation but a breakdown product of tire wear (Tian *et al*., 2020). Coho pre-spawn mortality is positively correlated with the relative proportion of roads, impervious surfaces, and commercial land cover within a basin (Feist et al. 2011). A model of the effects of pre-spawn mortality on coho salmon populations indicates that, depending on future rates of urbanization, localized extinction of coho salmon populations could occur within a matter of years to decades (Spromberg and Scholz 2001). Hopefully, tires can be re-formulated in time to prevent this. This finding emphasizes the significance of efforts to address both point-source and non-point-sources of contaminants in the landscape.

The following water quality subsections closely follow those provided in *Riparian Ecosystems, Volume 1: Science Synthesis and Management Implications* from WDFW (Quinn et al. 2020).

Sediment

Excessive sediment loads can significantly degrade water quality and, furthermore, sediments tend to serve as a transport mechanism for other pollutants, carrying attached contaminants from upland sources to the stream channel. Fine sediments that settle out of the water column can smother gravel and cobble streambeds that are essential habitat for salmonid spawning and for benthic macroinvertebrates. Suspended sediment can cause gill abrasion in fish and interfere with foraging and predator avoidance (Quinn et al. 2020).

Sediment input to streams is supplied by bed and bank erosion, landslides, and upland erosion processes. Other contaminants, including heavy metals and phosphorus, readily bind to suspended clay particles, and these contaminants are often transported with fine sediment in stormwater. Excess inputs of fine sediments into a stream channel reduce habitat quality for fish, amphibians, and macroinvertebrates. Fine sediment adversely affects stream habitat by filling pools, embedding gravels, reducing gravel permeability, and increasing turbidity. In salmon-bearing streams, fine sediment fills interstitial spaces in redds, reducing the flow of oxygenated water to developing embryos and reducing egg-to-fry survival (Jensen et al. 2009). Similarly, embedded, low-permeability gravel substrates reduce aquatic insect production, such insects being a primary food source for juvenile salmonid fish. Higher levels of fine sediment are also correlated with lower salmonid growth rates (Suttle et al. 2004). Highly turbid water can impair fertilization success in spawning salmonids (Galbraith et al. 2006) and interfere with the respiration and reproduction of amphibians (Knutson et al. 2004).

Vegetated riparian zones help stabilize stream banks and slow and filter overland flow, and temporarily store sediment that is gradually released to both seasonal and perennial streams. Sediment filtration is also high within intermittent and ephemeral streams, presumably because of the high interface with vegetative structures and the flux in water surface elevation, which allows for sediment storage along the streambanks (Dietrich and Anderson 1998).

Upland clearing and grading can result in long-term increases in fine sediment inputs to streams (Gomi et al. 2005, Jackson et al. 2007). Numerous studies have investigated the effectiveness of varying widths of buffers at filtering sediment. These studies have typically found high sediment filtration rates in relatively narrow buffer areas (Sheridan et al. 1999, reviewed in Wenger 1999, reviewed in Parkyn 2004, reviewed in Yuan et al. 2009) without a significant improvement in sediment retention beyond 15 meters (Abu-Zreigh et al. 2004).

It is significant to note, however, that field plot experiments tend to have much shorter field lengths (hillslope length contributing to drainage) than would be encountered in real-world scenarios (*i.e.*, ~5:1 ratio of field length to riparian width for a field plot compared to 70:1 ratio in NRCS guidelines). Since water velocities tend to increase with field length, field plot experiments may suggest better filtration than would be encountered under real-world conditions. Additionally, field-scale experiments generally do not account for flow convergence, which reduces sediment retention (Helmers et al. 2005) or for stormwater components that bypass filter strips through ditches, stormwater infrastructure, and roads (Verstraeten et al. 2006). Therefore, the effectiveness of filter strips at filtering sediment under real-world conditions and at the catchment scale is likely to be lower than what is reported in field plot experiments.

Additionally, many studies on sediment retention in riparian zones consider sediment retention from one storm event, rather than accounting for sediment accumulation over time. Two studies used Cesium-137 to track the location of sediment deposition over many years (Lowrance et al. 1988 and Cooper et al. 1988 in Wenger 1999). Together these studies suggest that riparian zones from 30-100 m (98-328 ft) or more may be necessary to provide long-term sediment retention, and that studies of short-term sediment retention underestimate the riparian zone width needed for ongoing sediment filtration.

In addition to width, the slope, vegetation density, and sediment composition of a riparian area have significant bearing on sediment filtration potential (Jin and Romkens 2001). A recent model of sediment retention in riparian zones found that a grass riparian zone as small as 4 m (13 ft) could trap up to 100% of sediment under specific conditions (2% hillslope over fine sandy loam soil), whereas a 30 m (98 ft) grass riparian zone would retain less than 30% of sediment over silty clay loam soil on a 10% hillslope (Dosskey et al. 2008) (Figure 2). This study exemplifies the effects that soil type and hillslope have on sediment retention.

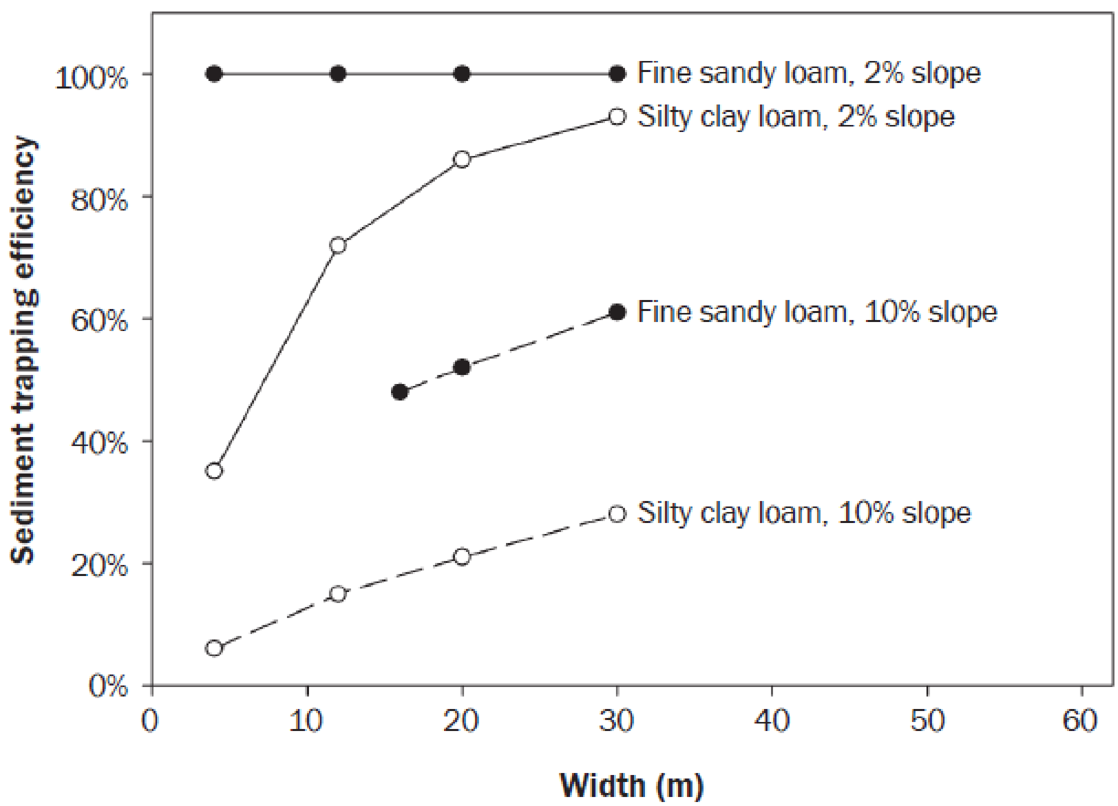

Figure 2. Sediment trapping efficiency related to soil type, slope, and buffer width. (Figure from Dosskey et al. 2008).

Multiple studies have found that larger particles tend to settle out within the first 3-6 m (10-20 ft) of the riparian zone, but finer particles that tend to degrade instream habitat, such as silt and clay, need a larger riparian zone, ranging from 15-120 m (49-394 ft), for significant retention (reviewed in Parkyn 2004).

Vegetative composition within the buffer also affects sediment retention. Vegetation tends to become more effective at sediment and nutrient filtration several years after establishment for both grass and forested buffers (Dosskey et al. 2007). Thin-stemmed grasses may become overwhelmed by overland flow while dense, rigid-stemmed vegetation provides improved sediment filtration that is expected to continue to function better over successive storm events (Blanco-Canqui et al. 2004, Yuan et al. 2009).

Nutrients

In excess concentrations, nitrogen and phosphorus can lead to poor water quality conditions, including reduced dissolved oxygen rates, increased pH, and eutrophication (Mayer et al. 2005, Mayer et al. 2007)). Excessive amounts of nitrogen and phosphorus speed up eutrophication and algal blooms in receiving waters, which can deplete the dissolved oxygen in the water and result in poor water quality and fish kills (Mayer et al. 2005, Dethier 2006, Heisler et al. 2008).

Riparian zones can reduce nitrogen pollution through nutrient uptake, assimilation by vegetation, and through denitrification (Sobota et al. 2012). The rate of nitrogen removal from runoff varies considerably depending on local conditions, including soil composition, surface versus subsurface flow, riparian zone width, riparian composition, and climate factors (Mayer et al. 2005, Bernal et al. 2007, Mayer et al. 2007). Nutrient assimilation is also dependent on the location of vegetation relative to the nitrogen source, the flowpath of surface runoff, and position in the landscape (Baker et al. 2006).

Nutrients enter waterways through channelized runoff, groundwater flow, and overland flow. Nitrogen loading is often associated with agricultural activities, whereas low density residential development has been found to result in nitrate levels comparable to a forested basin (Poor and McDonnell 2007).

Mayer et al. (2005, 2007) found that there was little relationship between riparian zone width and removal of *subsurface* nitrates. Subsurface nitrates were removed effectively regardless of riparian zone width. However, nitrate removal from *surface* runoff *was* related to riparian zone width, where 50%, 75%, and 90% surface nitrate removal was achieved at widths of 27 m (88 ft), 81 m (266 ft), and 131 m (430 ft) respectively (Mayer et al. 2007). This suggests that surface water infiltration in the riparian zone should be a priority to promote effective nutrient filtration. Where soils are poorly drained and infiltration capacity is limited, the effectiveness of nutrient removal in riparian buffers may also be limited (Wigington et al 2003).

The size and species composition of the riparian zone buffer also affects the efficiency of nutrient removal, but studies are conflicting as to whether grass, wetland, herbaceous, or forested buffers are most effective at removing nutrients (reviewed in Polykov 2005). Where nitrogen-fixing species predominate, such as red alder, these buffers tend to have higher soil nitrate concentrations (Monohan 2004).

Removal of phosphorus by riparian buffers is dependent on the form of phosphorus entering the buffer. Whereas phosphorus that is adsorbed by soil particles is effectively removed through sediment retention within a buffer, the retention of soluble phosphorus relies on infiltration and uptake by plants (Polyakov et al. 2005). One long-term study found that phosphorus uptake was directly proportional to the plant biomass production and root area over the four-year study period (Kelly et al. 2007). If a riparian buffer becomes saturated with phosphorus, its capacity for soluble phosphorus removal will be more limited (Polyakov et al. 2005). Another long-term study found that following a 15-year establishment period, a 40-meter (131 ft) wide, three-zoned buffer reduced particulate phosphorus by 22 percent, but dissolved phosphorus exiting the buffer was 26 percent higher than the water entering the buffer, so the buffer resulted in no net effect on phosphorus (Newbold et al. 2010).

In summary, most riparian zones reduce subsurface nutrient loading, but extensive distances are needed to reduce nutrients in surface runoff. Filtration capacity decreases with increasing loads (Mayer et al. 2005), so best management practices across the landscape that reduce nutrient loading will improve riparian function.

Metals

Although most metals can be toxic at high concentrations, cadmium, mercury, copper, zinc, and lead are particularly toxic even at low concentrations. Chronic and acute exposure to heavy metals have been found to impair, injure, and kill to aquatic plants, invertebrates, fish, and particularly salmonids (Grant and Ross 2002, Dethier 2006, Hecht et al. 2007, McIntyre et al. 2008, McIntyre et al. 2012). A review of contaminant effects on aquatic organisms summarized the factors affecting the toxicity of metals as follows:

- Duration and concentration of exposure
- The form of the metal at the time of exposure
- Synergistic, additive or antagonistic interactions of co-occurring contaminants
- Species sensitivity
- Life stage
- Physiological ability to detoxify and/or excrete the metal and,
- The condition of the exposed organism (ESV Environment Consultants 2003).

Metals are typically transported to the aquatic environment through fossil fuel combustion, industrial emissions, municipal wastewater discharge, and surface runoff (ESV Environment

Consultants 2003). In general, heavy metals and hydrocarbons are found in road runoff, and these contaminants can reach the City's streams directly through existing stormwater systems. Stormwater systems that circumvent buffers limit the opportunity to filter runoff through adjoining soils and vegetation. Accordingly, stream buffers are typically underutilized for treatment of metals, hydrocarbons, and other pollutants found in typical stormwater runoff.

Pathogens

While not necessarily a problem for fish and other wildlife, waterborne pathogens associated with human and animal wastes are a concern for direct and indirect human exposure. Although pathogens include a suite of bacteria and viruses, fecal coliform bacteria is typically used as an indicator of the possible or presumed presence of these pathogens. Fecal pollution tends to be positively correlated with human population densities and impervious surface coverage (Glasoe and Christy 2004). The main sources of fecal pollutants include municipal sewage systems, on-site sewage systems, stormwater runoff, marinas and boaters, farm animals, pets, and wildlife (Glasoe and Christy 2004). As municipal wastewater systems have improved treatment quality and capacity in recent years, increasingly, non-point source (septic systems, stormwater, wildlife, and pets) pollution is responsible for fecal contaminants in surface water (Glasoe and Christy 2004).

Within the City, the lower reaches of Issaquah Creek are on the state's 303(d) list of impaired waters for temperature and dissolved oxygen (WDOE, electronic source). The 303(d) list includes waters classified as Category 5 for various pollutants, which indicates "Polluted water that requires a water quality improvement project." Such a water quality project is typically an established Total Maximum Daily Load (TMDL). In addition, Lewis, Tibbetts, and North Fork Issaquah Creeks have been assessed as Category 5 for temperature and dissolved oxygen, and Lewis and Laughing Jacobs Creeks have been assessed as Category 5 for bacteria. Ongoing monitoring is required as part of TMDLs.

Herbicides and Pesticides

Commonly used herbicides and pesticides may also affect aquatic communities, and the acute and chronic effects of these chemicals or combinations of chemicals are not always well understood. Additionally, effects documented in the laboratory may differ significantly from effects identified in a field setting (Relyea 2005, Thompson et al. 2004). Despite our limited understanding, the effects of these chemicals may be long-lasting, as has been observed for legacy pesticides such as polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) in salmon, seabirds, and marine mammals in Puget Sound (Calambokidis et al. 1984, O'Neill et al. 1998, Ross et al. 2000, Wahl and Tweit 2000, Grant and Ross 2002, O'Neill et al. 2009).

Herbicides and pesticides may reach aquatic systems through a number of pathways, including surface runoff, erosion, subsurface drains, groundwater leaching, and spray drift. Narrow hedgerows have been found to limit 82-97 percent of the aerial drift of pesticides adjacent to a stream (Lazzaro et al. 2008). In runoff, herbicide retention in a buffer is dependent on the percentage of runoff that infiltrates the soil (Misra et al. 1996). A study of herbicides in simulated runoff found that 6-meter-wide vegetated buffers were sufficient to reduce herbicide concentration exiting the buffer to zero (Otto et al. 2008). A meta-analysis found that filtration effectiveness increased logarithmically from 0.5 m to an asymptote at approximately 18 m (Zhang et al. 2010). In summary, relatively narrow vegetated buffers may be effective in limiting herbicides and pesticides from reaching aquatic habitats in surface runoff, erosion, and spray drift; however, transport via subsurface drainage and leaching are not affected by riparian buffers, and these processes are best managed through the use of best management practices in herbicide and pesticide applications to avoid contaminating groundwater (Reichenberger et al. 2007).

Pharmaceuticals

Pharmaceuticals are another class of contaminants, the effects of which remain poorly understood. Many commonly used pharmaceuticals are found in wastewater, particularly around more urban areas (Long et al. 2013). Many common pharmaceuticals have endocrine-disrupting properties, which can affect fertility and development in non-target aquatic species (Caliman and Gavrilescu 2009). The existing and potential population-scale effects of these chemicals in the environment are not yet well-understood (Mills and Chichester 2005, Caliman and Gavrilescu 2009).

Wildlife Habitat

Riparian ecosystems, including streams and associated riparian areas, including wetlands, provide important wildlife habitat within the landscape due to the presence of unique structures and processes. Ecological features that are linked to species richness and abundance in a landscape include structural complexity, connectivity to other ecosystems, plentiful sources of food and water, and a moist moderate microclimate (Knutson and Naef 1997). Riparian ecosystems, depending on site-specific conditions, landscape position, and surrounding land use, will have some or all of these habitat features.

These aquatic ecosystems provide habitat for a broad range of fauna including invertebrates, reptiles and amphibians, anadromous and resident fish, birds, and mammals. Aquatic invertebrates that depend on stream and wetland ecosystems are important to aquatic trophic systems or food webs (Rosenberg and Danks 1987, Wissinger 1999, both in Sheldon et al. 2005). Native frogs and salamanders require wetlands for breeding. Buffer condition, habitat interspersion, wetland hydro-period, and diameter of emerged plant stems are all important factors that impact amphibian richness and abundance (Sheldon et al. 2005). Wetlands with surface connections to salmon-bearing streams can provide backwater refuge for anadromous fish if they also have ponded water at least 18 inches deep, low flow conditions, and cover such as overhanging or submerged plants (Sheldon et al. 2005). Waterfowl rely upon riparian ecosystems for all or part of their life cycle (Kauffman et al. 2001, in Sheldon 2005). Suitability of habitat for birds is dependent on buffer condition and width, presence of snags or other perches, corridor connections, open water, and forest canopy cover (Sheldon et al. 2005). Mammals typically associated with water, such as beaver and muskrat, also seek out well buffered vegetated corridors, interspersed habitat with open water, and a seasonally stable water level (Sheldon et al. 2005). According to a Washington Department of Fish and Wildlife (WDFW) study conducted by Knutson and Naef (1997) a predominance of terrestrial vertebrate species in Washington are dependent on streams and riparian areas, including wetlands.

Riparian ecosystems also provide habitat for many native plants species. Wetland characteristics that are correlated with plant richness are the hydro-period, duration of flooding, and variety of water depths (Schueler 2000 and Sheldon et al. 2005). Vegetated areas surrounding streams and wetlands perform several important functions that in turn protect their habitat functions.

Habitat fragmentation is a consequence of urbanization. As land is developed, continuous tracts of native habitat are reduced to patches, which become progressively smaller and more isolated. Dale et al. (2000) found that ecologic impacts of development are often overlooked and landscape-scale changes, particularly habitat fragmentation, alter the structure and function of those ecosystems.

The performance of stream and wetland habitat functions is affected to varying degrees by the width and/or character of the surrounding buffers. Urbanization reduces wetland buffering and increases human encroachment. Disturbance vectors include noise; nighttime light; physical intrusion by equipment, people, or pets; and garbage. Each of these vectors can result in one or more of the following: disruption of essential wildlife activities, damage to native vegetation and invasion of non-native species, erosion or fill, among others. Semlitsch and

Bodie (2003) found that upland areas surrounding wetlands are core habitats for many semi-aquatic species, such as amphibians and reptiles.

Cumulative impacts of direct and indirect riparian ecosystem alterations, including hydrologic changes, compromised water quality, and habitat fragmentation tend to reduce the habitat functions and values an urban wetland provides (Sheldon et al. 2005, Azous and Horner 2010).

Protection Measures

For the purpose of this discussion, clarification needs to be made between the terms "stream buffers" and "riparian management zones" (RMZs). WDFW current guidance (Windrope et al 2020) for protection of riparian areas heavily emphasizes a shift in terminology from the concept of "stream buffers" to "riparian management zones" (RMZs). An RMZ is defined as "…a scientifically based description of the area adjacent to rivers and streams that has the potential to provide full function based on the SPTH [site potential tree height] conceptual framework." This differs from the use of "buffer(s)," as an RMZ is by definition wide enough to potentially provide full riparian function. Stream buffers are established through policy decisions and are clearly intended to protect streams, but may or may not be intended to provide full riparian function or a close approximation of it.

As noted above, habitat types in the City of Issaquah vary across the landscape. The city falls mainly within the Puget Trough Ecoregion, though eastern portions of the city lie within the West Cascades Ecoregion. Both of these ecoregions tend to be forested except in areas cleared for agriculture or development or, for the West Cascades Ecoregion, in alpine areas above tree line in elevation and climate. No portions of the more arid ecoregions, such as are found east of the Cascades, are present in the City. These arid ecoregions have only sparse and/or scattered forested areas. The Puget Trough Ecoregion is forested primarily with Douglas fir (Pseudotsuga menziesii), western red cedar (Thuja plicata), and western hemlock (Tsuga heterophylla), and also includes black cottonwood (Populus trichocarpa) and red alder (Alnus rubra) along river channels more subject to disturbance. In the West Cascades Ecoregion, the same tree species occur at lower elevations with added Pacific silver fir and noble fir at mid elevations and stands of mountain hemlock and subalpine fir in the highest areas (Landscope Washington, 2022, http://www.landscope.org/washington/natural_geography/ecoregions/). The plant community and ecology of this region has important implications for how riparian buffers are established or RMZs defined.

WDFW's current recommendations for establishing RMZ widths are based primarily on a Site Potential Tree Height (SPTH) framework. The SPTH of an area is defined as "…the average maximum height of the tallest dominant trees (200 years or more) for a given site class." Exceptions may occur where SPTH is less than 100 feet, in which case WDFW recommends assigning an RMZ width of 100 feet at a minimum based primarily on what is needed to provide adequate biofiltration and infiltration of runoff for water quality protection, but also in consideration of other habitat-related factors including shade and wood recruitment (Windrope et al. 2020). WDFW recommends measuring RMZ widths from the outer edge of the Channel Migration Zone (CMZ), where present, or from the Ordinary High Water Mark (OHWM) where a CMZ is not present.

To apply their methodology, WDFW has developed a web-based mapping tool for use in determining SPTH in forested ecoregions of the state, such as occur in Issaquah. Where SPTH is 100 feet or more, WDFW recommends full protection within one SPTH, driven by the largest dominant tree species at any location.

Below is a graphical representation of the Forest Ecosystem Management Assessment Team (FEMAT) curves, the same as or similar to that included in WDFW's recommendations for establishing the bounds of RMZs (Windrope et al 2020). The curves show percentage of full function for various riparian habitat attributes with increasing distance from a stream. The WDFW recommendations show this graphic and these curves to justify recommending one full SPTH for RMZ width to attain "full" riparian function. However, an examination of the graphed habitat functions reveals that most of them top out somewhat or well before reaching a distance from the channel of one full SPTH. This is clearly true for root strength, shading, and litter fall as graphed. The possible exception is coarse wood recruitment, but even there only the final few percentage points are added for roughly the last 25% of the distance, beyond approximately 0.75 SPTH. And where old-growth conditions do not already exist within a buffer or RMZ, up to 200 years would be needed for this added small percentage of large wood to actually accrue. The curves indicate that nearly all of the benefits of RMZ function are realized at a distance of 70-80% of SPTH, with very small improvements beyond that.

In fact, a very high level of performance can be achieved with a width somewhat less than a full SPTH, as is demonstrated by the FEMAT curves. Arguably, some additional small gains would be realized even beyond a distance of 100% SPTH. The SPTH is really just one point along a continuum of diminishing returns, with the highest rates of return on habitat function generally occurring at and near the streambank and diminishing from there with distance.

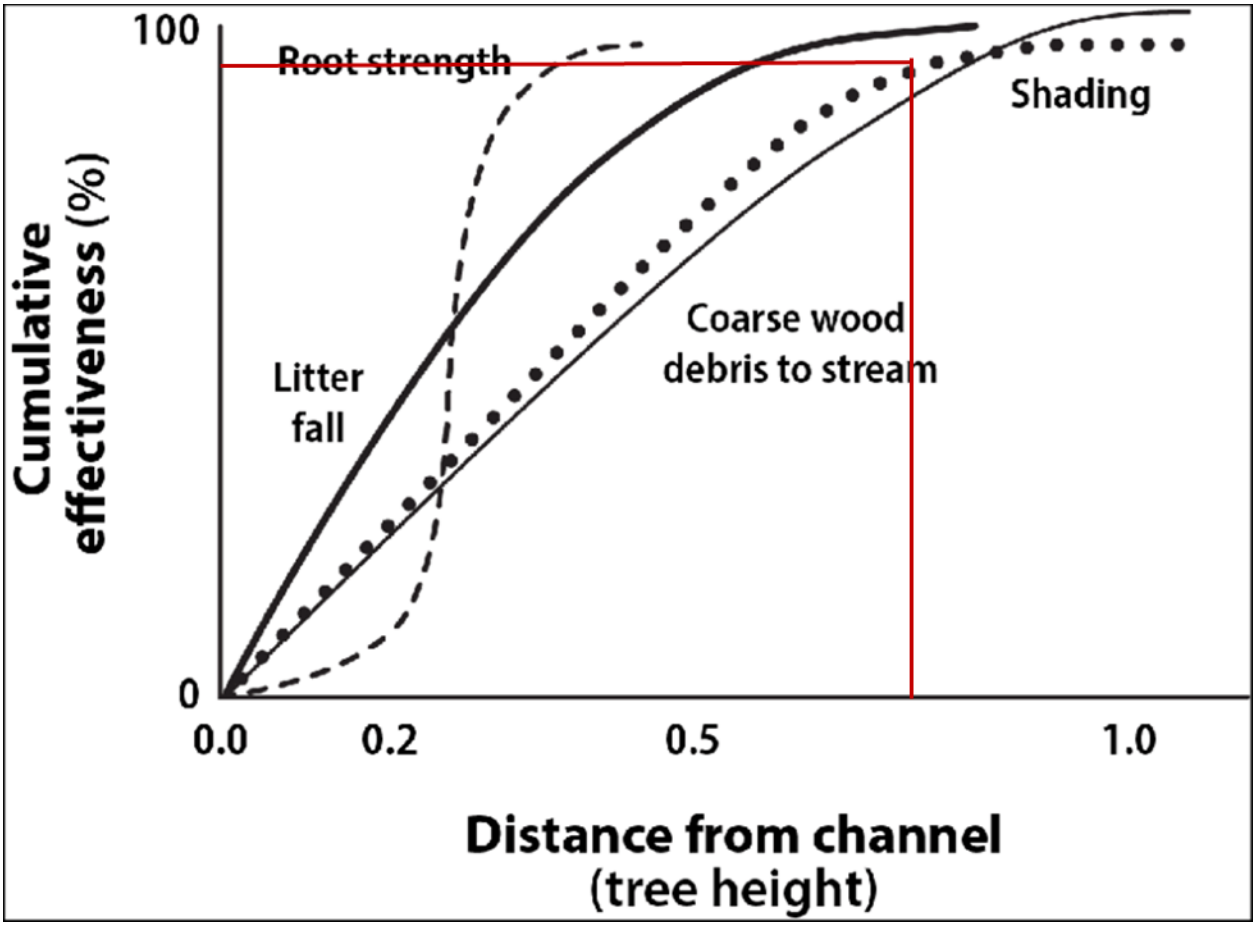

Figure 3. The "FEMAT Curves" (FEMAT 1993): a generalized conceptual model describing contributions of key riparian ecosystem functions to aquatic ecosystems as the distance from a stream channel increases. "Tree height" refers to average height of the tallest dominant tree (200 years old or greater); referred to as site-potential tree height (SPTH).

The list below is not an exhaustive evaluation of the relevant data, but we used WDFW's on-line mapping tool and found the following ranges of values for SPTH in feet for various dominant forest types throughout the City of Issaquah, Douglas-fir being the predominant species, and red alder present to some extent:

Douglas-fir	196-231 feet
Red Alder	100-111 feet

This informal sampling indicates that the riparian buffer widths proposed in the draft CAO for Type F streams (100 feet), tend to be on the low end of the SPTH range for Red Alder, and well below the range for Douglas Fir. Considering the WDFW guidance and BAS review, The

Watershed Company recommends increasing the Type F stream buffer to 150 feet. 150 feet falls above the SPTH range for Red Alder but below the range for Douglas Fir, this is a good compromise or approximation for assigning a fixed width rather than a custom delineation for buffer or RMZ widths. We feel that these values support a conclusion that a 150-foot Type F riparian buffer is a good approximation for the widths that would be determined by the recommended SPTH mapping process for determining RMZ widths. Some widths determined using the SPTH mapping process are above and some below a 150-foot buffer width. Both methods result in a high level of riparian habitat protection and a 150-foot buffer takes into consideration typical SPTH values for the City. As can be seen from the FEMAT curves exhibit, above, a buffer width slightly below a full SPTH results in only a small, near-undetectable reduction in habitat function.

Proposed Code Updates

The City's CAO currently classifies streams as a separate critical area type, rather than including them as a Fish and Wildlife Habitat Conservation Area (FWHCA) as defined by WAC 365-190-130. Furthermore, the CAO uses the old Class 1-4 stream classification system, rather than the updated Washington State Water Typing system (WAC 222-16-030). Class 2 streams are currently divided into two categories, those with salmonid use and those without, with each having their own corresponding buffer width, for a total of 5 buffer width categories ranging from 25 to 100 feet. The proposed Title 18 amendments will change this and incorporate streams as FHWCAs with a classification system consistent with the WAC 222-16-030. This will result in the elimination of the fifth category of streams, effectively increasing the lowest stream buffer from 25 to 50 feet for the lowest category of streams, which will now be classified as Type Ns. Similarly, the two categories of Class 2 streams will be combined into the Type F class, effectively increasing the buffer width on some streams from 75 to 100 feet. The proposed stream types and corresponding buffer widths in the current draft are depicted in Table 1 below.

Table 1. Stream buffer table from 2-10-22 draft CAO

Table 18.802.320.F Required Stream Buffer Widths

Stream Type	Required Buffer (feet)
S	100
F	100
Np	75
Ns	50

Note that the Type S stream buffers were evaluated and approved under the Shoreline Master Program (SMP) adoption process in 2021.

Additionally, the current update proposes a revision to the stream definition, and eliminates the buffer reduction and buffer averaging provisions, to better align with the BAS demonstrating that full buffer widths are needed to provide adequate riparian function and stream protection.

As described above, considering the WDFW guidance and BAS reviews, The Watershed Company recommends establishing a Type F buffer width of 150 feet. Therefore, we recommended replacing the stream buffer table in the current draft CAO with Table 2, below.

Table 2. TWC Proposed Standard Riparian Buffers (Measured horizontally from and perpendicular to OHWM)

Stream Type	Standard Buffer (feet)
S	See SMP
F	150
Np	75
Ns	50

A 150-foot Type F riparian buffer is a good approximation for the widths that would be determined by the recommended SPTH mapping process for determining RMZ widths, and would better align with BAS than the current proposal.

References

Author. 2004. City of Issaquah Best Available Science Report.

Abu-Zreig, M., R.P. Rudra, M.N. Lalonde, H.R. Whiteley, and N.K. Kaushik. 2004. Experimental Investigation of Runoff Reduction and Sediment Removal by Vegetated Filter Strips. Hydrological Processes. 18: 2029-2037. Published online 12 May 2004 in Wiley InterScience (www.interscience.wiley.com). DOI: 10.1002/hyp.1400.

Azous, A. and Horner, R. 2010. Wetlands and urbanization: implications for the future. CRC Press.

Baker M.E., D.E. Weller, and T.E. Jordan. 2006. Improved methods for quantifying potential nutrient interception by riparian buffers. Landscape Ecol 21(8):1327-45.

Bernal, S., F. Sabater, A. Butturini, E. Nin, and S. Sabater. 2007. Factors limiting denitrification in a Mediterranean riparian forest. Soil Biology & Biochemistry 39 (10): 2685-2688.

Beschta, R. L., Bilby, R. E., Brown, G. W., Holtby, L. B., and Hofstra, T. D. 1987. Stream temperature and aquatic habitat: Fisheries and forestry interactions. Pages 191-232 *in* E. O. Salo, and T. W. Cundy, editors. Streamside management: Forestry and Fishery Interactions. University of Washington, Seattle, WA.

Bilby, R.E. 1981. Role of organic debris dams in regulating the export of dissolved and particulate matter from a forested watershed. Ecology 62:1234-1243. (i)

Bisson, P.A., R.E. Bilby, M.D. Bryant, C.A. Dollof, G.B. Grette, R.A. House, M.L. Murphy, K.V. Koski, and J.R. Sedell. 1987. Large woody debris in forested streams in the Pacific Northwest: past, present, and future. Pages 143-190 in E.O. Salo, and T.W. Cundy, editors. Streamside management: forestry and fishery interactions. Contribution No. 57. Institute of Forest Resources, University of Washington, Seattle. (viii)

Blanco-Canqui, H., C.J. Gantzer, S.H. Anderson, and E.E. Alberts. 2004. Grass barriers for reduced concentrated flow induced soil and nutrient loss. Soil Science Society of America Journal 68:1963-1972.

Bolton, S. and J. Shellberg. 2001. Ecological Issues in Floodplains and Riparian Corridors. Report to Washington Dept. of Transportation, Olympia.

Bragg, DC. 2000. Simulating Catastrophic and Individualistic Large Woody Debris Recruitment for a Small Riparian System. Ecology 81(5):1383-1394.

Brosofske, K.D., J.Q Chen, R.J. Naiman, and J.F. Franklin. 1997. Harvesting effects on microclimatic gradients from small streams to uplands in Western Washington. Ecological Applications 7: 1188-1200.

Bury, R.B. 2008. Low thermal tolerances of stream amphibians in the Pacific Northwest: implications for riparian and forest management. Applied Herpetology 5:63-74. (i)

Calambokidis, J. et al. 1984. Chemical Contaminants in Marine Mammals from Washington State. NOAA Tech. Memo. NOS OMS 6.

Caliman, F.A. and M. Gavrilescu. 2009. Pharmaceuticals, personal care products and endocrine disrupting agents in the environment - a review. Clean Soil, Air, Water 37:4-5.

Castelle, A.J. and A.W. Johnson. 1998. Riparian vegetation effectiveness. *In* Abstracts from the Salmon in the City conference. Center for Urban Water Resources Management, University of Washington, 65 pp.

Dale, V. H., S. Brown, R. A. Hauber, N. T. Hobbs, N. Huntly, R. J. Naiman, W. E. Riebsame, M. G. Turner, and T. J. Valone. 2000. Ecological Principles and guidelines for managing the use of land. Ecological Applications 10: 639-670.

Dethier, M. 2006. Native Shellfish in Nearshore Ecosystems of Puget Sound. Puget Sound Nearshore Partnership Report No. 2006-04. Seattle District, U.S. Army Corps of Engineers, Seattle, WA.

Dietrich, W. E., & Anderson, N. H. (1998). Dynamics of Abiotic Parameters and Sediment Retention in Summer-dry Headwater Streams of Western Oregon. Hydrobiologia, 379(1-3), 1–15.

Dosskey, M.G., K.D. Hoagland, and J.R. Brandle. 2007. Change in filter strip performance over ten years. J Soil Water Conserv 62(1):21-32.

Dosskey, M. G., M. J. Helmers, and D. E. Eisenhauer. 2008. A design aid for determining width of filter strips. Journal of Soil and Water Conservation 63(4):232-241.

Dudley, S., J. C. Fischenich, and S. R. Abt. 1998. Effect of woody debris entrapment on flow resistance. J. Amer. Water Res. Assoc. 34(5): 1189-1197.

Fausch, K. D., and T.G. Northcote. 1992. Large woody debris and salmonid habitat in a small coastal British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences 49:682-693.

Feist, B. E., E.R. Buhle, P. Arnold, J.W. Davis and N.L. Scholz. 2011. Landscape Ecotoxicology of Coho Salmon Spawner Mortality in Urban Streams. PLoS ONE 6(8):e23424. doi:10.1371/journal.pone.0023424.

FEMAT (Forest Ecosystem Management Assessment Team). 1993. Forest ecosystem management: An ecological, economic, and social assessment. U.S. Departments of Agriculture, Commerce, and Interior. Portland, Oregon.

Fleeger, J. W., K.R. Carman, and R.M. Nisbet. 2003. Indirect Effects of Contaminants in Aquatic Ecosystems. The Science of the Total Environment.

Galbraith, R.V., E.A. MacIsaac, J. Macdonald, J. Stevenson, A.P. Farrell. 2006. The effect of suspended sediment on fertilization success in sockeye (Oncorhynchus nerka) and coho (Oncorhynchus kisutch) salmon. Canadian Journal of Fisheries & Aquatic Sciences.

Glasoe, S. and A. Christy. 2004. Literature Review and Analysis: Coastal Urbanization and Microbial Contamination of Shellfish Growing Areas. Puget Sound Action Team. Publication #: PSAT04-09

Gomi, T., D. Moore, and M. Hassan. 2005. Suspended sediment dynamics in small forest streams of the pacific northwest. J Am Water Resour Assoc 41(4):877-98.

Gomi, T., R. D. Moore, and A.S. Dhakal. 2006. Headwater Stream Temperature Response to Clear-cut Harvesting with Different Riparian Treatments, Coastal British Columbia, Canada. Water Resources Research. Vol. 42, W08437, doi:10.1029/2005WR004162.

Grant, S.C.H and P.S. Ross. 2002. Southern Resident Killer Whales at Risk: Toxic Chemicals in the British Columbia and Washington Environment. Canadian Technical Report of Fisheries and Aquatic Science 2412. Fisheries and Oceans Canada.

Grizzel, J., M. McGowan, D. Smith, and T. Beechie. 2000. Streamside buffers and large woody debris recruitment: evaluating the effectiveness of watershed analysis prescriptions in the North Cascades region. TFW-MAG1-00-003. Olympia, WA. Timber Fish and Wildlife 37p.

Gurnell, A.M., H. Piegay, F.J. Swanson, and S.V. Gregory. 2002. Large wood and fluvial processes. Freshwater Biology, 47(4), 601–619.

Harmon, M.E., J.F. Franklin, F.J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, S.P. Cline, N.G. Aumen, J.R. Sedell, G.W. Lienkaemper, K. Cromack Jr., and K.W. Cummins. 1986. Ecology of Coarse Woody Debris in Temperate Ecosystems. Advances in Ecological Research 15:133-301.

Hecht, S.A., D.H. Baldwin, C.A. Mebane, T. Hawkes, S.J. Gross, and N.L. Scholz. 2007. An overview of sensory effects on juvenile salmonids exposed to dissolved copper: applying a benchmark concentration approach to evaluate sublethal neurobehavioral toxicity. NOAA Technical Memorandum NMFS-NWFSC-83.

Heisler, J., M. Glibertb, J.M. Burkholder, D.M. Anderson, W. Cochlane, W.C. Dennison, Q. Dortch, C.J. Gobler, C.A. Heil, E. Humphries, A. Lewitus, R. Magnien, H.G. Marshall, K. Sellner, D.A. Stockwell, D.K. Stoecker, M. Suddleson. 2008. Eutrophication and harmful algal blooms: A scientific consensus. Harmful Algae 8 (3-13)

Helmers, M.J., D.E. Eisenhauer, M.G. Dosskey, T.G. Franti, J.M. Brothers, and M.C. McCullough. 2005. Flow pathways and sediment trapping in a field-scale vegetative filter. Transactions of the ASAE 48:955-968.

Jackson, C. R., D.P. Batzer, S.S. Cross, S.M. Haggerty, C.A Sturm.. 2007. Headwater streams and timber harvest: Channel, macroinvertebrate, and amphibian response and recovery. Forest Science.

Jensen, D.W., E.A. Steel, A.H. Fullerton, G.R. Pess. 2009. Impact of Fine Sediment on Egg-To-Fry Survival of Pacific Salmon: A Meta-Analysis of Published Studies. Reviews in Fisheries Science.

Jin, C.X. and M.J.M. Romkens. 2001. Experimental studies of factors in determining sediment trapping in vegetative filter strips. Trans. ASAE 44:277-288.

Johnston, N. T., S.A. Bird, D.L. Hogan, and E.A. MacIsaac. 2011. Mechanisms and source distances for the input of large woody debris to forested streams in British Columbia, Canada. Canadian Journal of Forest Research, 41(11), 2231–2246.

Kelly J.M., J.L. Kovar, R. Sokolowsky, T.B. Moorman. 2007. Phosphorus uptake during four years by different vegetative cover types in a riparian buffer. Nutr Cycling Agroecosyst 78(3):239-51.

King County Department of Natural Resources. 2006. Preliminary Feasibility Analysis for the Lewis Creek Fish Passage Improvement at the Spooner Property.

King County Surface Water Management Division, Department of Public Works. October 1991. Issaquah Creek Basin Current/Future Conditions and Source Identification Report.

King County SWM. 1996. Final Issaquah Creek basin and nonpoint action plan. King County Surface Water Management. Seattle, Washington.

Knutson, K. L., & Naef, V. (1997). Management recommendations for Washington's priority habitats: riparian. Washington Department of Fish and Wildlife.

Knutson, M.G., W.B. Richardson, D.M. Reineke, B.R Gray, J.R. Parmelee, and S.E. Weick. 2004. Agricultural ponds support amphibian populations. Ecological Applications 14:669-684.

Lazzaro, L., Otto, S., and G. Zanin. 2008. Role of hedgerows in intercepting spray drift: Evaluation and modelling of the effects. Agriculture, Ecosystems and Environment, 123(4):317-327; Feb 2008

Lienkaemper, G.W., and F.J. Swanson. 1987. Dynamics of large woody debris in streams in old-growth Douglas-fir forests. Canadian Journal of Forest Research 17:150-156.

Long, E. R., M. Dutch, S. Weakland, B. Chandramouli and J.P. Benskin. 2013. Quantification of pharmaceuticals, personal care products, and perfluoroalkyl substances in the marine sediments of Puget Sound, Washington, USA. Environmental Toxicology and Chemistry, 32: 1701–1710.

May, C. L., and R.E. Gresswell. 2003. Large wood recruitment and redistribution in headwater streams in the southern Oregon Coast. Canadian Journal of Forest Research, 33, 1352–1362. doi:10.1139/X03-023.

Mayer, P.M., S.K. Reynolds, D. McCutchen, and T.J. Canfield. 2005. Riparian Buffer Width, Vegetative Cover, and Nitrogen Removal Effectiveness: A Review of Current Science and Regulations. EPA/600/R-05/118. Cincinnati, Ohio, U.S. Environmental Protection Agency.

Mayer, P.M., S.K. Reynolds, J. Marshall, D. McCutchen, and T.J. Canfield. 2007. Meta- Analysis of Nitrogen Removal in Riparian Buffers. Journal of Environmental Quality. 36: 1172-1180.

McDade M.H., F.J. Swanson, W.A. McKee, J.F. Franklin, and J. Van Sickle . 1990. Source distances for coarse woody debris entering small streams in western Oregon and Washington. Can J Forest Res 20:326–330.

McIntyre, J. K., D. H. Baldwin, J. P. Meador, and N. L. Scholz. 2008. Chemosensory deprivation in juvenile coho salmon exposed to dissolved copper under varying water chemistry conditions. Environmental Science & Technology 42:1352-1358.

McIntyre, J. K., D.H. Baldwin, D. Beauchamp, and N.L. Scholz. 2012. Low-level copper exposures increase visibility and vulnerability of juvenile coho salmon to cutthroat trout predators. Ecological applications : a publication of the Ecological Society of America, 22(5):1460–71.

Mills, L.J. and C. Chichester 2005. Review of evidence: Are endocrine-disrupting chemicals in the aquatic environment impacting fish populations? Sci Total Environ 343(1-3):1-34.

Misra, A.K., J.L. Baker, S.K. Mickelson, and H. Shang. 1996. Contributing area and concentration effects on herbicide removal by vegetative buffer strips. Trans. ASAE 39:2105-2111.

Monohan, C.E. 2004. Riparian buffer function with respect to nitrogen transformation and temperature along lowland agricultural streams in Skagit County, Washington. Dissertation, Univ. Washington, Seattle.

Moore, A.A. and M.A. Palmer. 2005. Invertebrate biodiversity in agricultural and urban headwater streams: Implications for conservation and management. Ecological Applications 15:1169-1177.

Moore, R.D. and S.M. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: a review. Journal of the American Water Resources Association 41:763-784

Murphy, M. L. and K. V. Koski. 1989. Input and depletion of woody debris in Alaska streams and implications for streamside management. North American Journal of Fisheries Management.9(4):427-436.

Murray, G. L. D., R. L. Edmonds, and J. L. Marra (2000), Influence of partial harvesting on stream temperatures, chemistry, and turbidity in forests on the western Olympic Peninsula, Washington, Northwest Sci., 74, 151– 164.

Naiman, R.J., C.A.Johnson, and J.C. Kelley. 1988. Alteration of North American Streams by Beaver: structure and dynamics of streams are changing as beaver recolonize their historic habitat. Bioscience 38(11)753-762.

Nakamura, F. and F. Swanson. 1993. Effects of coarse woody debris on morphology and sediment storage of a mountain stream system in western Oregon. Earth Surface Processes and Landforms 18: 43-61.

Newbold, J. D., S. Herbert, B.W. Sweeney, P. Kiry, and S.J. Alberts. 2010. Water Quality Functions of a 15-Year-Old Riparian Forest Buffer System. JAWRA Journal of the American Water Resources Association, 46: 299–310.

O'Neill, S.M., J.E. West, and J.C. Hoeman. 1998. Spatial Trends in the Concentration of Polychlorinated Biphenyls (PCBs) in Chinook (Oncorhynchus tshawytscha) and Coho Salmon (O. kisutch) in Puget Sound and Factors Affecting PCB Accumulation: Results from the Puget Sound Ambient Monitoring Program. Washington Department of Fish and Wildlife.

O'Neill, , and Sandra M. James E. 2009. West. Marine Distribution, Life History Traits, and the Accumulation of Polychlorinated Biphenyls in Chinook Salmon from Puget Sound, Washington. Transactions of the American Fisheries Society, Volume 138, 2009 - Issue 3, Pages 616-632 | Received 09 Jan 2008, Accepted 11 Dec 2008, Published online: 09 Jan 2011

Otto S., M. Vianello, A. Infantino, G. Zanin, and A. Di Guardo. 2008. Effect of a full-grown vegetative filter strip on herbicide runoff: Maintaining of filter capacity over time. Chemosphere 71(1):74-82.

Parametrix. 2003. Stream Inventory and Habitat Evaluation Report Including Issaquah Creek, East and North Forks of Issaquah Creek, Tibbetts Creek, and the Shoreline of Lake Sammamish

Parkyn, S. 2004. Review of Riparian Buffer Zone Effectiveness. Canada Ministry of Agriculture and Forestry (MAF). Technical Paper No: 2004/05.

Pollock, M., G. Pess, T. Beechie, D. Montgomery. 2004. The importance of beaver ponds to coho salmon production in the Stillaguamish River basin, Washington, USA. American Journal of Fisheries Management 24: 749-760.

Polyakov, V. A. Fares, and M.H. Ryder. 2005. Precision Riparian Buffers for the Control of Nonpoint Source Pollutant Loading into Surface Water: A Review. Environmental Review. 13: 129-144. Published on the NRC Research Press Web site at http://er.nrc.ca/ on 16 August 2005.

Poole, G.C. and C.H. Berman. 2001. An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. Environmental Management 27(6): 787-802.

Poor, C. and J. McDonnell. 2007 . The effects of land use on stream nitrate dynamics. J Hydrol (Amst) 332(1-2):54-68.

Quinn, T., G.F. Wilhere, and K.L. Krueger, technical editors. 2020. Riparian Ecosystems, Volume 1: Science Synthesis and Management Implications. Habitat Program, Washington Department of Fish and Wildlife, Olympia, Washington.

Reichenberger, S., M. Bach, A. Skitschak, and H.G. Frede. 2007. Mitigation strategies to reduce pesticide inputs into ground- and surface water and their effectiveness; a review. Sci Total Environ 384(1-3):1-35.

Relyea, R.A. 2005. The lethal impact of Roundup® on aquatic and terrestrial amphibians. Ecological Applications 15:1118-1124.

Robison, E.G. and R.L. Beschta. 1990. Identifying trees in riparian areas that can provide coarse woody debris to streams. Forest Science. 36(3):790-801

Roni, P. and T.P. Quinn. 2001. Density and size of juvenile salmonids in response to placement of large woody debris in western Oregon and Washington streams. Canadian Journal of Fisheries & Aquatic Sciences.

Ross, P.S., G.M. Ellis, M.G. Ikonomou, L.G. Barrett-Lennard and R.F. Addison. 2000. High PCB Concentrations in Free-Ranging Pacific Killer Whales, Orcinus orca: Effects of Age, Sex and Dietary Preference. Marine Pollution Bulletin. Vol. 40, No. 6: 504-515.

Scholz, N. L., M. S. Myers, S. G. McCarthy, J. S. Labenia, J. K. McIntyre, G. M. Ylitalo, L. D. Rhodes, C. A. Laetz, C. M. Stehr, B. L. French, B. McMillan, D. Wilson, L. Reed, K. D. Lynch, S. Damm, J. W. Davis, and T. K. Collier. 2011. Recurrent die-offs of adult coho salmon returning to spawn in Puget Sound lowland urban streams. PloS one 6(12):e28013.

Schueler, T.R. 2000. The Impact of Stormwater on Puget Sound Wetlands. Technical Note #109 from Watershed Protection Techniques 3(2), Article 33.

Sedell, J.R., P.A. Bisson, F.J. Swanson, and S.V. Gregory. 1988. What we know about large trees that fall into streams and rivers. In Maser C., R.F. Tarrant, J.M. Trappe, and J.F. Franklin. From the Forest to the Sea: a Story of Fallen Trees. USDA Forest Service General Technical Report PNW-GTR-229.

Semlitsch, R.D. and J.R. Bodie. 2003. Biological Criteria for buffer zones around wetland and riparian habitats for amphibians and reptiles. Conserv. Biol. 17:1219-1228.

Sheldon, D., T. Hruby, P. Johnson, K. Harper, A. McMillan, T. Granger, S. Stanley, and E. Stockdale. 2005. Wetlands in Washington State, Vol. 1: A Synthesis of the Science. Washington State Department of Ecology Publication #05-06-006. Olympia, WA.

Sobota, D. J., S.L., Johnson, S.V. Gregory, and L.R. Ashkenas. 2012. A Stable Isotope Tracer Study of the Influences of Adjacent Land Use and Riparian Condition on Fates of Nitrate in Streams. Ecosystems 15:1-17

Sorrano, P.A., S.L. Hubler, S.R. Carpenter, and R.C. Lathrop. 1996. Phosphorus loads to surface waters: A simple model to account for spatial pattern of land use. Ecol. Appl. 6(3):865–878

Spromberg, J.A. and N.L. Scholz. 2011. Estimating the Future Decline of Wild Coho Salmon Populations Resulting from Early Spawner Die-Offs in Urbanizing Watersheds of the Pacific Northwest, USA. Integrated Env. Assessment and Management. Vol 7. No 4: 648-656.

Sridhar, V., A.L. Sansone, J. LaMarche, T. Dubin, and D.P. Lettenmaier. 2004. Prediction of Stream Temperature in Forested Watersheds. Journal of the American Water Resources Association (JAWRA) 40(1): 197-213.

Suttle, K. B., M. Power, J. Levine, and C. McNeely. 2004. How fine sediment in riverbeds impairs growth and survival of juvenile salmonids. Ecological Applications 14(4):969-974.

Swanson F.J., G.W. Lienkaemper, and J.R. Sedell. 1976. History, physical effects, and management implications of large organic debris in western Oregon streams. USDA Forest Service General Technical Report PNW-56.

Thompson, D.G., B.F. Wojtaszek, B. Staznik, D.T., and G.R. Stephenson. 2004. Chemical and biomonitoring to assess potential acute effects of Vision® herbicide on native amphibian larvae in forest wetlands. Environmental Toxicology and Chemistry 23: 843-849.

Tian, Z., H. Zhao, K. Peter, M. Gonzalez, C. Wu, J. Prat, E. Mudrock, R. Hettinger, A.E. Cortina, R.G. Biswas, F. C. Kock, R. Soong, A. Jeene, B. Du, F. Hou, H. He, R. Lundeen, A. Gilbreath, R. Sutton, N.L. Scholz, J. W. Davis, M.C. Dodd, A. Simpson, J.K. Mcintyre, and E. P. Kolodziej.. 2020. A ubiquitous tire rubber–derived chemical induces acute mortality in coho salmon. Science Volume 371, Issue 6525: 185-189. .

United States Environmental Protection Agency (USEPA). 2003. EPA Region 10 guidance for Pacific Northwest state and tribal temperature water quality standards. EPA 910-B-03-002. Region 10 Office of Water, Seattle, WA.

Van Sickle, J., and S. V. Gregory. 1990. Modeling inputs of large woody debris to streams from falling trees. Canadian Journal of Forest Research 20: 1593-1601.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37: 130-137.

Verstraeten, G, J. Poesen, K. Gillijns, G. Govers. 2006. The use of riparian vegetated filter strips to reduce river sediment loads: an overestimated control measure? Hydrol Process 20(20):4259-67.

Wahl, T. R., B. Tweit, S. Mlodinow, editors. 2005. Birds of Washington: status and distribution. Oregon State University Press, Corvalis, OR.

Watershed Company, The. 2006. Stream and Riparian Areas Restoration Plan.

Wenger, S. 1999. A review of the scientific literature on riparian buffer width, extent, and vegetation. Office of Public Service and Outreach, Institute of Ecology, University of Georgia. Internet: http://outreach.ecology.uga.edu/toos/buffers/lit_review.pdf.

Wigington, P.J. Jr, S.M. Griffith, J.A. Field, J.E. Baham, W.R. Horwath Owen, J.H. Davis, S.C. Rain and J.J. Steiner. 2003. Nitrate removal effectiveness of a riparian buffer along a small, agricultural stream in Western Oregon. Journal of Environmental Quality 32:162-170.

Windrope, A., T. Rentz, K. Folkerts, and J. Azerrad. 2020. Riparian Ecosystems, Volume 2: Management Recommendations: A Priority Habitats and Species Document of the Washington. Department of Fish and Wildlife, Olympia, Washington.

Wondzell, S. M., J. Lanier, et al. 2009. Changes in hyporheic exchange flow following experimental wood removal in a small, low-gradient stream. Water Resources Research 45(5).

WRIA 8 Salmon Recovery Council. 2017. Lake Washington/Cedar/Sammamish Watershed Chinook Salmon Conservation Plan 10-year Update (2017). Water Resources Inventory Area (WRIA) 8, Seattle, WA.

Wynn, T. and S. Motsaghimi. 2006. The effects of vegetation and soil type on streambank erosion, southwestern Virginia, USA. Journal of the American Water Resources Association 42(1):69-82.

Yuan, Y.P., R.L. Bingner, and M.A. Locke. 2009. A Review of effectiveness of vegetative buffers on sediment trapping in agricultural areas. Ecohydrology 2(3):321-336.

Zhang, X., X. Liu, M. Zhang, and R.A. Dahlgren. 2010. A review of vegetated buffers and an meta-analysis of their mitigation efficacy in reducing nonpoint source pollution. Journal of Environmental Quality 39:76-84.