

**The technical basis of zone of sensitivity determinations
under the detailed assessment procedure of
the Riparian Areas Regulation**

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Forward

The B.C. Ministry of Environment (MOE), and Fisheries and Oceans Canada (DFO) jointly prepared this report to document technical basis for determining the “Zones of Sensitivity” used in the *Detailed Assessment* procedure as part of the Riparian Areas Regulation. The RAR seeks to maintain riparian areas so that they may provide the natural features, functions and conditions that support fish life processes. The Detailed Assessment procedure employs a “process based” approach to determining riparian widths; this report is intended to explain the technical foundation for this approach.

This document synthesizes available information and identifies that there is still significant scientific uncertainty with respect to protecting riparian values while not unduly limiting development opportunities. The approach to implementing the RAR will explicitly address this uncertainty by systematically evaluating both compliance with, and the effectiveness of, this management approach. The evaluations will quantify how well the approach achieves the objective of protecting riparian features and functions and, if objectives are not being achieved, will form the basis for any future modifications to the RAR.

In addition to determining the “Zone of Sensitivity”, the *Detailed Assessment* also requires measures to protect the integrity of riparian areas. These measures are not discussed here but include: maintaining slope stability, preventing erosion and sediment transport during construction, developing and implementing stormwater management plans, maintaining the natural movement of streams in floodplains, protecting riparian trees, evaluating risks and mitigation options for hazard trees, addressing risks of windthrow, and preventing encroachment into riparian areas. A more detailed description of these measures can be found in the RAR Assessment Methodology.

MOE, DFO and the Union of B.C. Municipalities (UBCM) are committed to adaptively managing the Riparian Areas Regulation (RAR) through an Intergovernmental Cooperation Agreement. The RAR Steering Committee formed by the Intergovernmental Cooperation Agreement will draw upon these evaluations and others to guide the implementation of the RAR and to determine its future direction.

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Introduction

A purpose of the Riparian Areas Regulation (RAR) is to “establish directives to protect riparian areas from development so that the areas can provide natural features, functions, and conditions that support fish life processes” and so ensure that “there will be no harmful alteration, disruption or destruction of natural features, functions, and conditions that support fish life processes in the riparian assessment area” (B.C. Reg. 376/2004). Thus, the RAR attempts to manage riparian areas along streams and lakes in watersheds subject to urban development so as to maintain the quantity and quality of aquatic habitat at the levels that would occur in the absence of anthropogenic disturbance. Streams are, however, dynamic systems whose physical and biological characteristics vary spatially and temporally in response to changes in inputs of materials and energy. Stochastic natural variation (e.g., in discharge), whose impacts may be magnified by the effects of site disturbance on watershed processes such as sediment transport or large woody debris inputs (Hartman and Scrivener 1990), can profoundly alter local habitat conditions (Hogan et al. 1998). The downstream movement of water and materials connects, and may amplify, impacts from disturbances at different locations within a watershed. In urban watersheds, anthropogenic disturbances will occur at many sites and times. However, the regulatory tools to minimize the adverse impacts of riparian disturbance normally can be applied only at individual disturbance locations. The site assessment required under the RAR attempts to maintain the physical and biological processes that determine habitat characteristics and productivity. If ecosystem processes at all locations throughout the watershed operate “normally”, the RAR will maintain aquatic habitat conditions similar to those that would occur in the natural state. A regulatory approach to aquatic habitat management that is focussed on processes rather than specifying acceptable target levels for metrics of local aquatic habitat quantity or quality (e.g., Anonymous 2000, Fox and Bolton 2007) may circumvent technical difficulties associated with measuring and interpreting habitat conditions, detecting change in variable environments, and linking (undesirable) changes to specific disturbances, which are inherent in approaches that specify acceptable target values for habitat characteristics. A process-based approach can result in operational procedures that are technically simple, objectively and consistently applicable, and effective in achieving management goals if applied throughout the whole watershed.

The processes that determine the physical characteristics of stream habitats in forested watersheds are largely those that govern the input, storage, and transport of sediment and water. The processes that affect the biological productivity of stream habitats are primarily those that govern the input, storage, and transformation of energy in the form of organic matter, heat, and kinetic energy. The RAR directives, and similar regulations elsewhere (see Lee et al. 2004), attempt to avoid anthropogenic disturbances within riparian areas which might alter the rates of such processes, either directly (e.g., by increasing insolation at

a site) or indirectly by amplifying the effects of natural disturbance (e.g., by increasing sediment transport at peak flood flow). Clearly, only those processes whose rates are strongly influenced by conditions within riparian areas can be managed by riparian area regulations, but such processes can include insolation, litterfall, large woody debris delivery, overland flows, nutrient removal, and certain aspects of sediment delivery, whose effects on instream conditions are well established (Gregory et al. 1991, Murphy and Meehan 1991, Richardson and Danehy 2007).

Other influences on stream habitat that are less closely tied to riparian conditions (urbanization and watershed deforestation effects on water yield, water quality, and sediment routing, e.g., Moscrip and Montgomery 1997, Paul and Myer 2001, Konrad et al. 2005) must be mitigated through other means. Because the effectiveness of riparian management in maintaining desirable physical and biological states in urban streams can be reduced by watershed-level changes to hydrologic and sediment regimes (Roy et al. 2006, Walsh et al. 2007), watershed-scale planning to avoid or mitigate the effects of urbanization on stream hydrology and sediment routing is an essential prerequisite to effective riparian management. Riparian management cannot fully mitigate the adverse effects of urbanization on stream hydrology (Booth 2005), so effective water management planning must be implemented as urbanization proceeds in a watershed. Detailed B.C. guidelines for stormwater management at the watershed level are available (Anonymous 2002).

Streams in heavily-urbanized watersheds may also be managed for policy objectives (e.g., public safety, transportation infrastructure) in addition to the preservation of stream habitat values. The RAR assumes that managing for multiple objectives can be done in a manner that allows watershed processes to operate normally. Where this assumption is not met, active management of stream habitat may be required to maintain acceptable conditions.

The assumptions on which the RAR is based are thus: that maintaining undisturbed or restoring site-potential natural riparian vegetation of “sufficient” extent will, on average, maintain riparian-influenced biophysical processes at the same rates as in a undisturbed condition; that urban development beyond the undisturbed riparian area will not alter the characteristics of the riparian forest from those of the regulated state; and that effects on stream habitat which originate beyond the riparian forest will be mitigated. Experimental comparisons of stream habitat adjacent to logged, minimally disturbed, and undisturbed riparian forests of varying extents suggest that this approach can preserve desired instream conditions, at least over periods of a few years to a few decades (Murphy et al. 1986, Kreutzweiser et al. 2004, 2005, Wilkerson et al. 2006). A key issue within a RAR management framework is determining the

extent of the undisturbed riparian forest that is required in given circumstances to maintain unimpaired the rates of riparian-influenced biophysical processes throughout the cycle of stand development and replacement. Monitoring for compliance then becomes a simple matter of ensuring that the required extent is met and maintained. Monitoring for effectiveness forms part of a continuous cycle of evaluation and adaptation for the regulations.

Management approach

The detailed assessment procedure under the RAR identifies a “zone of sensitivity” (ZOS) adjacent to a waterbody within which disturbance is restricted, i.e., an undisturbed riparian buffer. The use of forested riparian buffers to maintain biophysical processes that affect stream habitat conditions is not novel: resource management agencies in at least 60 jurisdictions in North America use riparian buffers in regulations or guidelines designed to protect streams and other waterbodies (Lee et al. 2004). The effects of the biophysical processes through which riparian vegetation influences instream conditions vary continuously with buffer width, but eventually reach asymptotes beyond which there is little further incremental effect (FEMAT 1993 cited in Naiman et al. 2000, and Fig. 1). A buffer width that is at or

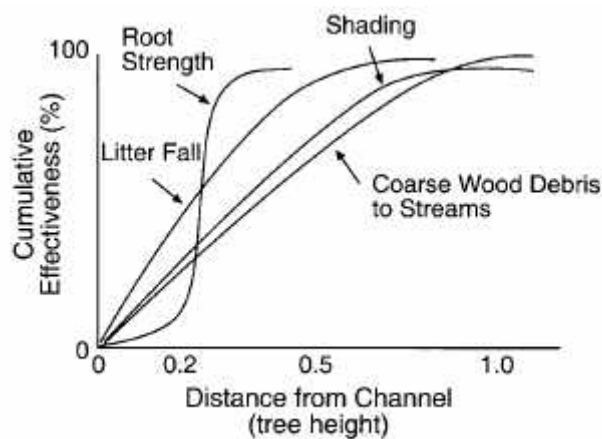


Figure 1. Conceptual model of the effect of forested buffers on processes that determine instream conditions. Note that effects vary with buffer width, but eventually reach asymptotes which differ among processes (after FEMAT 1993).

beyond the maximum asymptote for the set of processes being considered should maintain the instream conditions that are influenced by these processes at values near those of an undisturbed site. Thus, most agencies prescribe a fixed buffer width that is intended to maintain instream conditions similar to those found in undisturbed forests or suitable for target species such as salmonid fishes. Despite the universality of the underlying biophysical processes, required buffer widths are surprisingly variable among

jurisdictions (Young 2000, Lee et al. 2004). These differences may reflect the relative importance of different biophysical processes in regions with different climates, soils, and vegetations as well as different management goals. Additionally, many agencies recognize that site factors such as stream width or vegetation type alter the interactions between riparian areas and the stream, and therefore vary the prescribed fixed buffer width by stream width class, vegetation type, or slope class (Young 2000, Lee et al. 2004).

The RAR detailed assessment procedure explicitly considers the effects of site conditions on particular interactions between riparian areas and streams when defining a required zone of sensitivity. It identifies the zone of sensitivity as the maximum of those required to maintain the individual processes within desired conditions. This approach is adopted in part because the wide range of biogeophysical conditions that may be encountered within British Columbia results in different processes dominating impacts on stream habitat in different circumstances, and in part because empirical data has accumulated to suggest that the forested buffer widths required to maintain desired instream conditions vary with stream size and geomorphic setting. It is important to note that the RAR assessment is concerned solely with instream and littoral habitat conditions and their contributions to fish habitat; riparian buffers are used by management agencies to preserve many other resource values that require different, usually wider, buffer widths to be effective (Fischer and Fischenich 2000).

Scope of the zone of sensitivity determination

The zone of sensitivity determination of the detailed assessment procedure considers a limited set of processes that link instream or nearshore conditions to riparian conditions: inputs of particulate organic matter as litter or terrestrial insects, shading for temperature regulation, bank stability and local erosion, and the recruitment of large woody debris to the stream channel or nearshore area of lakes. It is designed for urban development at the habitat unit to reach scale along small- to medium-size streams, i.e., from the scale of the individual lot to large developments encompassing many lots. It also applies to shoreline development along small lakes and wetlands. It is primarily concerned with processes affecting the lateral transport of materials from the adjacent terrestrial habitat, and not longitudinal transport downstream. It does not consider directly the cumulative impacts of development within a watershed, which typically result in degradation of instream conditions once impervious surface area exceeds 5-20% (Finkenbine et al. 2000, Paul and Myer 2001, Booth et al. 2002). It does not directly consider water quality, hydrology, or sediment transport other than site impacts on infiltration, erosion and bank stability. Maintaining natural riparian vegetation will, of course, reduce the impacts of development on these values, but the ZOS is not designed to be the primary control on inputs of chemicals, stormwater discharge, or the overland transport

of sediments. It does not directly consider channel movement, which may be part of the normal evolution of the stream network; however, the required “measures identified ... as necessary to protect the integrity” of the ZOS may encompass channel migration. It assumes that the normal patterns of seral development will be maintained in riparian areas. It does not directly consider habitat fragmentation or barriers to dispersal that may result from the clustering of individual impact sites or from individual developments that encompass long lengths of stream. The omission of many of the above topics results from an inability to specify in advance where development will occur and what length of stream or lakeshore will be affected. These issues are best dealt with through watershed-scale planning and zoning and, where necessary, monitoring and active management.

Technical basis of zone of sensitivity determinations

Information that is pertinent to establishing zones of sensitivity for inputs of particulate organic matter as litter or terrestrial insects, shading for temperature regulation, bank stability and local erosion, and the recruitment of large woody debris to the stream channel or nearshore area of lakes is summarized in separate sections below. The format for each topic is to résumé briefly the importance of the process in modifying instream or nearshore lake habitat conditions, to outline current understanding of the operation of the process, to summarize recent empirical data that helps establish quantitative values to maintain “undisturbed” conditions, and to use this information to establish zones of sensitivity as functions of site characteristics. The concise reviews are not exhaustive, and are biased towards recent empirical data from B.C. or regions similar to areas of B.C. where the results might apply. Where topics have had recent comprehensive reviews, these are cited. Sections are self-contained, and have separate bibliographies.

The zone of sensitivity determinations recognize that the effects of processes linking riparian areas to stream or littoral habitats may vary continuously with site variables such as vegetation structure or stream size. Thus, the ZOS width for a process can vary between a minimum and a maximum value, depending on site-specific conditions (Table 1). This contrasts with the step-function approach used elsewhere in which a very small set of invariant buffer widths are applied to limited, broad classes of a site-specific modifier such as stream width or the presence or absence of fish (Young 2000, Lee et al. 2004). Minimum and maximum ZOS widths were estimated from available empirical studies as those widths consistently resulting in no detectable differences in habitat variables compared to intact forest, or those resulting in detectable but small differences which likely have little adverse impact on habitat conditions and fish production. Minimum and maximum ZOS widths spanned the range of site conditions, e.g., stream size, likely to be encountered in new urban developments in B.C. Some subjectivity is

Table 1. Zone of sensitivity determinations under the B.C. Riparian Area Regulations for: (a) shade, (b) litter and terrestrial insect inputs, (c) large woody debris inputs and bank stability for streams, and (d) large woody debris inputs and bank stability for lakes and wetlands. Vegetation height classes are defined as: low cover \leq 1 m < shrub \leq 5 m < trees.

(a) Shade

Vegetation type	Streams	Lakes and wetlands
Low cover	n/a	n/a
Shrub	2 x width (maximum 5 m)	5 m
Trees	3 x width (maximum 30 m)	30 m

(b) Litterfall and terrestrial insect inputs for streams, lakes, and wetlands

Vegetation type	Streams			Lakes and wetlands
	Zone of sensitivity	Minimum	Maximum	
Low cover	5 m	5 m	5 m	5 m
Shrub	2 x width	5 m	15 m	15 m
Trees	3 x width	10 m	15 m	15 m

(c) Large woody debris inputs and bank stability for streams

Channel Type^a	Vegetation type		
	Low cover	Shrub	Trees
Riffle-pool	3 times channel width to:		
	maximum of 5 m	maximum of 20 m	minimum of 10 m maximum of 30 m
Cascade-pool	2 times channel width to:		
	maximum of 5 m	maximum of 10 m	minimum of 10 m maximum of 15 m
Step-pool	1 times channel width to:		
	maximum of 5 m	maximum of 10 m	minimum of 10 m maximum of 10 m

(d) Large woody debris inputs and bank stability for lakes and wetlands

Vegetation type	Zone of sensitivity
Low cover	5 m
Shrub	5 m
Trees	15 m

^a Channel types follow the “Channel assessment procedure field guidebook” (Anonymous 1996) of the B.C. Forest Practices Code. Note that the CAP definitions of channel types may differ from similarly-named channel types under other widely-used classifications (e.g., Montgomery and Buffington 1997).

involved in summarizing available empirical data because there are usually insufficient data to map functional relationships between buffer widths and metrics of habitat response over the range of site factors known to influence them. Experimental comparisons of alternative buffer widths typically test only 2 or 3 discrete widths, but may also involve controlled disturbance (e.g., partial harvest) within the buffers to mimic particular management regimes. Synoptic surveys provide data over a broader range of buffer widths, but frequently confound buffer effects with other disturbance effects, e.g., by utilizing second growth sites whose vegetation and stream characteristics do not match those of wholly undisturbed sites, or by using forest plantations, or sites with partial harvest within the buffer. Differences among jurisdictions in prescribed riparian buffer widths for timber lands for similar site conditions (Young 2000, Lee et al. 2004) may partly reflect the subjectivity of interpreting ambiguous empirical information. In cases where data do not exist to allow empirical delineations of minimum or maximum ZOS widths to maintain desired conditions, these were estimated as “reasonable” values given current understanding of how the process operates. For example, given that litterfall models consistently estimate 90% dispersion distances that are less than the vegetation height, it seems unlikely that shrubs (defined as < 5 m height) will contribute litter from distances beyond the 15 m maximum ZOS.

Determining how ZOS widths should vary between minimum and maximum values as site factors change is problematic. There are few formal analyses of empirical data which determine buffer widths that will maintain a specified value of a metric of process effect as functions of the site variables that are believed to affect the process. Johnston et al. (2007) examined source distances for inputs of large woody debris to forested streams and found that source distances varied with vegetation height, stream size, and channel type, as expected, because the delivery mechanisms that recruited wood to the channel were correlated with these factors. In some cases, process models (e.g., RAIS, Welty et al. 2002) may provide guidance about the influence of site factors. For instance, RAIS suggests that the blockage of solar radiation inputs in simulated Douglas-fir forests was insensitive to buffer widths above about 15 m width. Although such models are best considered as working hypotheses rather than as accurate descriptions of interactions, they provide starting points for assessing the effects of site factors on some biophysical processes. In practice, continuously-varying factors whose effects are well-recognized but which are difficult to characterize are collapsed to broad categories within the RAR (e.g., three vegetation types, three channel types) but more easily determined site factors (e.g., stream width) are retained as continuous modifiers. The paucity of useable empirical data resulted in the variation in ZOS widths with stream size being approximated by simple multiples of stream width (1×, 2×, 3×) that matched the boundary conditions (minima and maxima) at appropriate stream widths. For example, using a 3× multiplier for determining a ZOS for large woody debris recruitment to treed riffle-pool channels resulted in the 10 m

minimum ZOS occurring at 3.3 m stream width and the 30 m maximum ZOS occurring at 10 m stream width. These ZOS widths are within the range of values applied elsewhere for such streams (see Young 2000, Blinn and Kilgore 2001, Lee et al. 2004). Because the operational ZOS width is specified as the maximum of those identified for the four biophysical processes considered within the detailed assessment for a site, uncertainties in defining the appropriate ZOS for a particular process may have little ultimate effect.

The RAR assessment procedures have been designed to be part of an adaptive management process in which the effectiveness of current procedures in maintaining natural processes will be actively monitored. The technical basis of ZOS determinations will change as new information becomes available, both from experimental comparisons of alternative management regimes and from monitoring the effectiveness of current ZOS designations in achieving policy goals. Procedures may be changed in the future to improve their effectiveness or ease of use.

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Inputs of leaf litter and terrestrial insects from riparian vegetation to streams

Streams in forested watersheds may receive considerable inputs of particulate organic matter (POM) as leaf litter and wood from the adjoining riparian vegetation (Benfield 1997, Johnston et al. 2003, Richardson et al. 2005). In very small forested streams, where shading by the riparian canopy limits instream primary production (Lamberti and Steinman 1997), inputs from the riparian forest may dominate the organic matter budget of the stream (Webster and Meyer 1997) and can be the dominant food source for aquatic invertebrates that are food for fish. Inputs of leaf litter from the riparian forest may greatly exceed algal production even on medium-size (15 m) forested streams when nutrient concentrations are low enough to limit algal production (Johnston et al. 2004). As channel size increases, however, the relative importance of instream primary production generally increases (Vannote et al. 1980, Connors and Naiman 1984, Doucett et al. 1996, Findlay 2001). Invertebrates in streams are believed to be adapted, both functionally and phenologically, to the dominant organic matter source (Vannote et al. 1980, Hawkins and Sedell 1981). Inputs of leaf litter from riparian forests thus strongly influence invertebrate production in small and medium-size forested streams (Wallace et al. 1999, Melody and Richardson 2004). Changes to particulate organic matter inputs from riparian forests alter both the composition and abundance of the stream invertebrates (Carlson et al. 1990, Webster et al. 1990, Davies and Nelson 1994, Kiffney et al. 2003). Small headwater streams also export particulate organic matter and invertebrates to downstream fish-bearing reaches (Webster et al. 1990, Wipfli and Gregovich 2002, Wipfli et al. 2007). The density and composition of the riparian vegetation influences the magnitude of this downstream export (Wipfli and Musslewhite 2004). Consequently, modifications of the riparian forest which alter organic matter inputs to streams may generate effects that extend well beyond the disturbed location. Fish production in small forested streams is highly dependence on terrestrial inputs (Doucett et al. 1996, Hicks 1997, Perry et al. 2003) but is less so in large streams (Doucett et al. 1996, Findlay 2001).

Riparian vegetation also exports terrestrial invertebrates to small streams (Wipfli 1997, Kawaguchi and Nakano 2001, Baxter et al. 2005). The abundance and taxonomic composition of terrestrial invertebrates on the nearbank vegetation varies among plant species (Mason and Macdonald 1982), and is greater for deciduous species than for conifers (Allan et al. 2003, Romero et al. 2005). Inputs of terrestrial invertebrates to streams “appear highest from closed-canopy riparian zones with deciduous vegetation” (Baxter et al. 2007, p. 201), but associations between inputs and riparian vegetation type and density are quite variable (e.g., Edwards and Huryn 1996, Kawaguchi and Nakano 2001, Allan et al. 2003, Romero et al. 2005, Romaniszn et al. 2007). The abundance and biomass of terrestrial invertebrate inputs and drift is greatest in summer (Wipfli 1997, Allan et al. 2003, Romero et al. 2005, Romaniszn et al. 2007), as is their consumption by stream-resident salmonids (Wipfli 1997, Kawaguchi and Nakano 2001).

Drifting terrestrial invertebrates may be preferentially selected by fish over aquatic invertebrates because of their generally larger size (Laudon et al. 2005), and terrestrial invertebrates may comprise a large portion of the diet of salmonids (Wipfli 1997, Kawaguchi and Nakano 2001, Allan et al. 2003). Some studies have found decreased fish biomass or production in non-forested reaches of small streams where terrestrial invertebrate inputs were reduced (Kawaguchi and Nakano 2001, 2003), while others have found increased fish densities and biomass in forested reaches compared to grassland reaches but no differences in fish production (Dineen et al. 2007).

The timing, quantity, and quality of leaf litter entering small streams depend on vegetation type and density (Richardson 1992, Scarsbrook et al. 2001). Deciduous leaves may dominate direct litterfall inputs to small and medium-size streams in some conifer-dominated forests (Connors and Naiman 1984, Richardson 1992, Johnston et al. 2003), but not invariably (Bilby and Bisson 1992, Richardson 1992). Litter inputs to small and medium-size streams increase with canopy closure (Scarsbrook et al. 2001, Johnston et al. 2003), but inter-site variation is high (Fig. 1). Unsurprisingly, the removal of riparian forest greatly reduces the quantity of litter inputs (Hartman and Scrivener 1990, Bilby and Bisson 1992, Kreutzweiser et al. 2004), and may replace conifer needle inputs by more-easily processed deciduous leaves (Bilby and Bisson 1992). Clearcutting old-growth Douglas-fir/western hemlock forest reduced

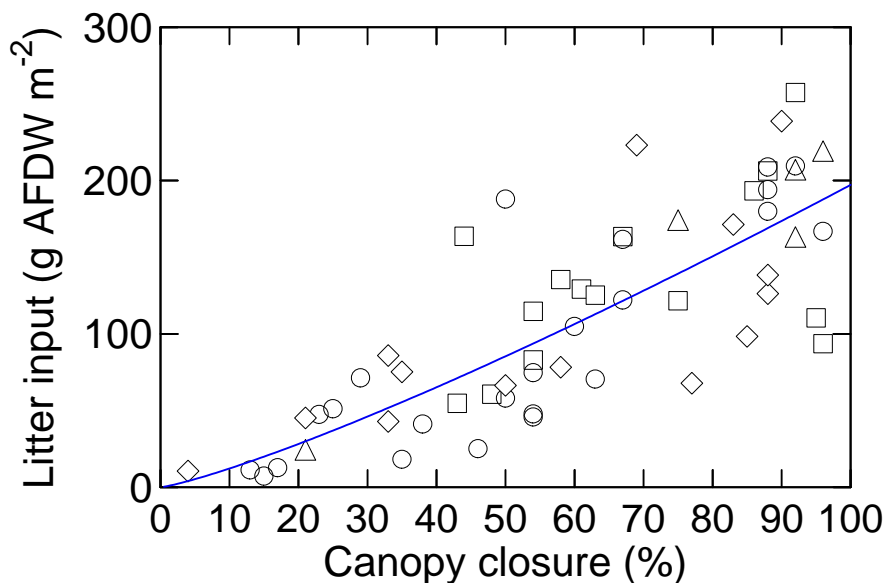


Fig. 1. Litter inputs increased with canopy closure at four streams (bankfull widths 8 m to 15 m) in undisturbed old-growth spruce/fir forest in north central B.C. Data are for individual litter collectors within the channels; symbols indicate different streams. (Johnston et al. 2003).

allochthonous inputs to about 20% of that of undisturbed forest (Bilby and Bisson 1992). Removal of all merchantable timber and most minor timber to the bank edge reduced deciduous litter inputs to 26% and coniferous inputs to 0% of values for undisturbed western hemlock/red cedar/Douglas-fir forest in the Carnation Creek, B.C. watershed (Hartman and Scrivener 1990). Partial harvest or similar disturbance to riparian vegetation may (Hartman and Scrivener 1990) or may not (Kreutzweiser et al. 2004) reduce litter inputs, presumably depending on forest type, the extent of the disturbance, and how it was applied. Variable-width riparian leave strips (1 m to 70 m) at Carnation Creek reduced inputs of litter from deciduous vegetation to 35% and conifer litter inputs to 27% of that of undisturbed forest (Hartman and Scrivener 1990). Selective harvest of up to 42% basal area within riparian zones did not alter litter inputs in mixed hardwood forest (Kreutzweiser et al. 2004). Reduced litter inputs may persist for many years afterward (Webster et al. 1990). The conversion of riparian forest to grassland will reduce terrestrial invertebrate inputs (Edwards and Hury 1996, Kawaguchi and Nakano 2001); however, the recolonization of harvested or disturbed riparian zones by deciduous species such as alder may result in little difference in the inputs of terrestrial invertebrates to the stream (Wipfli 1997, Piccolo and Wipfli 2002).

Riparian buffers along small streams may maintain normal inputs of terrestrial litter to streams and can maintain normal communities of forest-floor invertebrates within the near-stream riparian zone (Rykken et al. 2007). However, few regulations specifically consider buffer widths needed to maintain normal litter inputs, and there are few empirical data which examine the distances within the riparian zone from which litter recruits to a stream or the effectiveness of different buffer widths in maintaining terrestrial litter or invertebrate inputs to streams. Fischer and Fischenich (2000) recommended buffer widths of 3 m to 10 m to maintain litter inputs. Richardson et al. (2002) state that litter inputs from undisturbed 10 m and 30 m riparian buffer zones along small streams in southwestern B.C. did not differ from undisturbed controls. Grady (2001) found no relationship between riparian buffer widths and litter inputs to headwater streams in the western Cascade Mountains of Washington. Rykken et al. (2007) found that 30 m buffers preserved the distinct riparian forest-floor invertebrate community in close association with streams, as did dense shrub cover along a clearcut headwater stream.

Leaf litterfall models (Ferrari and Sugita 1996, Staelens et al. 2003, Jonard et al. 2006) and measurements of the decline in areal litter inputs with distance from the bank edge across stream channels (Connors and Naiman 1984, Johnston et al. 2003) provide information about distances from which leaf litter may enter stream channels. Measurements of the spatial dispersion of litterfall beneath isolated trees (15 m to 30 m height) in mixed hemlock/hardwood forests indicated that litterfall declined exponentially

with distance from the tree and that litter from deciduous trees dispersed further than litter from hemlock (Ferrari and Sugita 1996). Modelled dispersion indicated that 90% of the litter fell within 17 m; the 90% dispersal distances were slightly less than the tree heights. Similarly, modelled wind-influenced litterfall from hardwood forests indicated 75% dispersal distances of 11-26 m and 90% distances of 15-38 m (Staelens et al. 2003). A third model indicated 90% dispersal distances of 18-21 m downwind for mixed hardwood forests and 7-11 m upwind (Jonard et al. 2006); distances increased with tree height. Measured declines of litter inputs across streams suggest shorter recruitment distances. Connors and Naiman (1984) found that litterfall fell to near zero at distances of 8-10 m from the banks of a 30 m boreal forest river, and at 15-30 m on a 130 m wide river. Johnston et al. (2003) observed that deciduous leaf litter and total litter inputs declined exponentially with distance from the bank edge, and reached near zero at distances of 5-7 m on 8-15 m wide streams in old-growth spruce/fir forests (tree heights about 35 m). Inputs of conifer needles did not decline with distance, suggesting recruitment from longer distances, but comprised only a small portion of total inputs.

The empirical results from the B.C. buffer width experiment and from measurements of across-stream variation in litter inputs together suggest that riparian buffer widths of about 10-20 m will provide near-normal inputs of leaf litter to small streams in conifer forests. Exponential declines of litter inputs with distance from source mean that additional buffer width will have increasing smaller effects on total inputs. For example, integrating the relationship found by Johnston et al. (2003) over buffer widths of 5, 10, 15, and 20 m gives relative inputs 0.95, 0.99, 0.99, and 1 compared to a 30 m buffer width. Thus, underestimates of required buffer widths at treed sites may have little impact on the recruitment of litter to a stream. The zone of sensitivity determination to maintain inputs of litter and terrestrial invertebrates under the RAR adopts a minimum value of 10 m for treed sites, primarily on the basis of the buffer width experiment reported by Richardson et al. (2002). ZOS width increases with stream size to a maximum width of 15 m. Litter production and litter dispersal increase with tree height (Ferrari and Sugita 1996, Jonard et al. 2006), and the proportion of the total litterfall within a given distance increases rapidly as tree height decreases (Jonard et al. 2006). Modelled 90% dispersal distances were less than or similar to tree heights (Ferrari and Sugita 1996, Jonard et al. 2006). Therefore minimum ZOS widths for litter inputs are reduced to 5 m for shrubs (height \leq 5 m) and low cover (height \leq 1 m). The ZOS widths are increased to a maximum of 15 m for shrubs as stream size increases because of uncertainty over recruitment distances and processes, e.g., the effect of greater average wind speeds for more open systems, and the absence of empirical data to guide width determinations. Buffer widths are increased as stream width increases because the (limited) empirical data suggest greater litter dispersion distances at larger, more open rivers (Connors and Naiman 1984). The 2 \times and 3 \times width scalars for shrub and treed sites are untested, but

should encompass most litter source areas if the exponential declines in litter dispersal with distance from source determined by Ferrari and Sugita (1996) and Johnston et al. (2003) are approximately correct.

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Inputs of leaf litter and terrestrial insects from riparian vegetation to lakes

Small low-productivity lakes may obtain significant proportions of their carbon and nutrient budgets from terrestrial sources (Cole et al. 1990, Pace et al. 2004, Cole et al. 2006). Particulate organic matter (POM) from terrestrial sources can supply large proportions of the carbon flow to zooplankton and fish (e.g., 33-73% and 20-50% respectively, Cole et al. 2006). Litterfall from riparian vegetation provides a carbon subsidy to small unproductive lakes (Rau 1976, Richey and Wissmar 1979, France and Peters 1995), and pollen from forested watersheds can be a significant phosphorus subsidy to boreal lakes (Graham et al. 2006). Terrestrial prey, such as insects, can provide fishes with > 20% of their carbon (Cole et al. 2006) and can also be a major phosphorus input to small lakes (Cole et al. 1990).

Much of the terrestrial POM is deposited within about 10 m of the shore (Gasith and Hasler 1976, Rau 1976, Cole et al. 1990, France and Peters 1995) and originates as airborne litterfall from the nearshore vegetation, although some may also enter through overbank transfers (France 1995). Inputs reflect the composition of the adjacent nearshore vegetation. Rau (1976) found greatly decreased inputs adjacent to meadow and talus areas compared to those with coniferous forest at a subalpine lake, which suggests that low cover provides relatively small inputs of litter. Jones and Momot (1981) found that litter inputs were proportional to an index of total foliage abundance for the nearshore vegetation. In northwestern Ontario, most of the litterfall (primarily from conifers) entered during September and October (France and Peters 1995). France and Peters (1995, p.1581) found that “airborne litter input per offshore distance increased in relation to tree girth, proximity of trees to the lakeshore, and the elevation change in canopy height within 10 m inshore”. Litter inputs also increased with the percentage of the shoreline that was forested (France and Peters 1995). Above about 25% deforested shoreline, litter inputs declined to about 40-50% of their values at high forestation (France and Peters 1995).

Although none of these data specifically identifies the riparian distance from which plant litter was recruited to the nearshore lake environment, the rapid decline in inputs with distance offshore strongly suggests that this plant material does not move much further than 10 m, and so likely originates over a similar distance. Inputs appear to originate from the adjacent shore vegetation and vary with the type and abundance of the shore vegetation. The fixed ZOS widths of 5 m, 10 m, and 15 m for low cover, shrub, and treed site potential vegetation likely encompass source areas for leaf litter around small lakes while accounting for the strong inverse dependence on tree size observed by France and Peters (1995).

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Riparian vegetation and stream temperature

Riparian forests often influence water temperatures in adjacent small streams by intercepting and reducing direct inputs of solar radiation to the streams through absorption and reflection (Fig. 1). In many

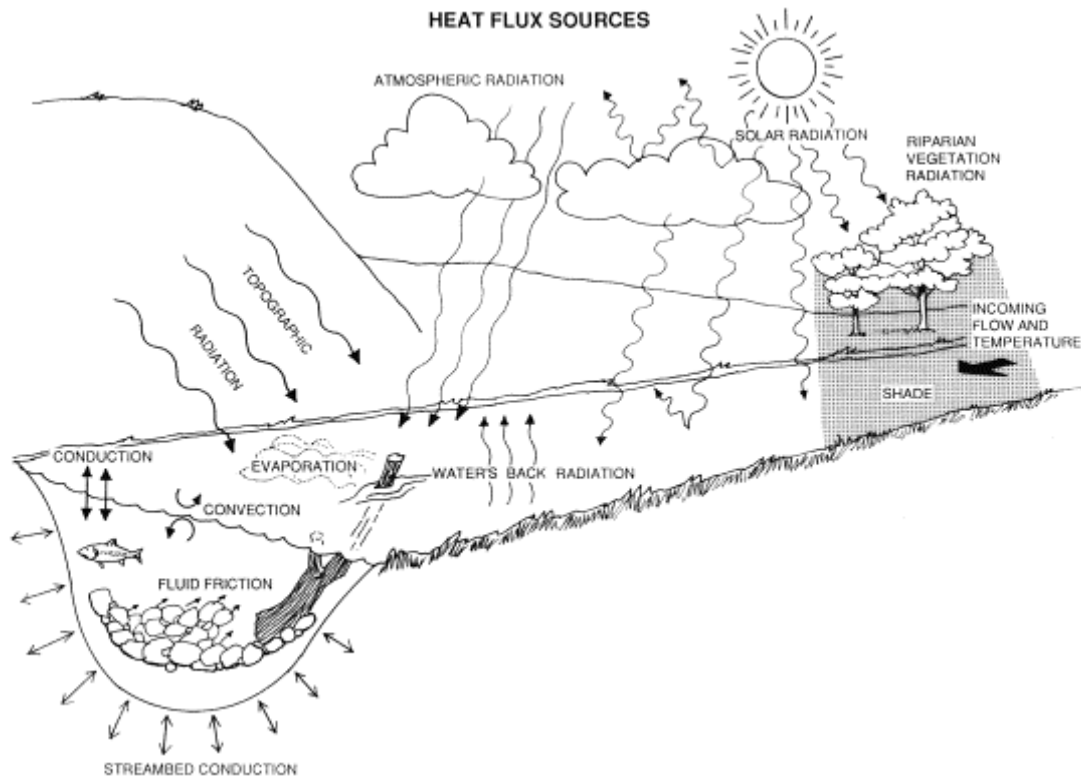


Fig. 1. Processes influencing the heat budget of a stream segment. Riparian vegetation alters the heat flux to the stream by reducing inputs from solar radiation (from Bartholow 2002).

cases, the heat budgets of small streams can be dominated by inputs via direct insolation, i.e., net short-wave radiation, in the absence of shading (Brown 1969, Johnson 2004, Webb and Zhang 2004, Moore et al. 2005, Cassie 2006). Shading by riparian vegetation reduces heat inputs and usually results in lower and less variable water temperatures (e.g., Kiffney et al. 2003, Macdonald et al. 2003, Webb and Crisp 2006, Wilkerson et al. 2006). The effect on stream temperature of shading by riparian vegetation will vary with factors such as: the geographic location of the site; season; channel orientation; topographic shading; vegetation type, density, height, and distribution; and stream reach length, width, water depth, and velocity, which jointly determine the heat flux via insolation and the volume of water to which it is added (Moore et al. 2005). Riparian vegetation can also influence other components of the heat budget of streams (Brosfokske et al. 1997, IMST 2000), but generally such effects are considerably less than impacts

on solar heating (Webb and Zhang 2004). Effects of shading can be large. The removal of riparian forests along small streams has increased mid-summer maximum stream temperatures by up to 12° C in some cases (Beschta et al. 1987, Moore et al. 2005). Groundwater inputs and hyporheic exchanges may, however, reduce the effects of solar heating on stream temperatures (Johnson 2004, Cassie 2006). Because temperature influences the rates both of abiotic chemical reactions and of physiological processes in poikilotherms, even small changes in stream temperature patterns can induce complex and significant effects on stream ecosystems (Hartmann and Scrivner 1990, p. 106 ff.). For stream-rearing fishes, temperature-induced effects may include altered rates of egg incubation and survival, inappropriate hatch timing, altered rates of growth and survival, and changes in abundance, distribution, and species composition (Hartmann and Scrivener 1990, Hicks et al. 1991).

Maintaining natural riparian vegetation along forested streams is often an effective mechanism to limit instream temperature changes that may result from anthropogenic disturbances such as timber harvest (Gomi et al. 2006, Wilkerson et al. 2006) or urban development (LeBlanc et al. 1997), and so to maintain natural temperature regimes. The technique is effective because stream temperatures respond to small-scale spatial variations in riparian shading (Burton and Likens 1973, Rutherford et al. 2004). It is also operationally simple to implement and monitor. Consequently, retaining riparian vegetation to maintain suitable levels of stream shading is frequently used by resource management agencies to maintain desired temperature characteristics in small forested streams (Young 2000). Where the management goal is to maintain near normal temperature regimes, as in the RAR, regulations are usually designed to maintain the natural characteristics of that portion of the riparian vegetation that contributes (or will contribute) to stream shading over the period of stand development. Alternatively, some agencies specify target levels of shading to achieve temperature characteristics that are suitable for stream-rearing fishes of management interest (Anonymous 2000). In either case, it is essential that catchment-level water management plans maintain normal stream hydrographs and sediment routing to avoid changes in the quantity of water to which solar heat is added or in the area of stream surface through which the flux of solar radiation enters, either of which will alter instream temperature (LeBlanc et al. 1997).

Management agencies also commonly require riparian buffers along lakeshores and wetlands (Young 2000, Lee et al. 2004). Direct shading by vegetation can be important in maintaining normal temperatures in shallow wetlands and in nearshore areas of small lakes (Steedman et al. 2001). Removal of shoreline forest along small Ontario lakes increased littoral summer maximum temperatures by several degrees (Steedman et al. 2001). If, however, littoral temperatures are locally influenced by groundwater inflows, riparian buffers may not maintain normal temperatures unless they encompass the groundwater

recharge zones that supply the inflows; terrestrial recharge zones may commonly be upslope of riparian areas (Buttle 2002). Although small increases in water temperatures can decrease growth and greatly increase mortality in lake populations of coldwater-adapted fish such as rainbow trout (Biro et al. 2007), logging around boreal lakes did not alter littoral fish community structure or abundance (Steedman 2003, Tonn et al. 2003). The removal of shoreline forest may also alter thermocline depths in small lakes by increasing midlake wind speed (France 1997) although Steedman et al. (2000) did not observe such an effect.

Undisturbed riparian buffer widths of roughly 10 m or more appear to maintain normal mid-summer temperatures for small (2 to 4 m) streams in coniferous forests (Gomi et al. 2006, Wilkerson et al. 2006), or to limit increases to 1-2 °C (Kiffney et al. 2003). For small streams, temperature regimes for undisturbed 30 m buffers are statistically indistinguishable from those of 10 m buffers (Kiffney et al. 2003, Gomi et al. 2006, Wilkerson et al. 2006), suggesting that shading effects arise primarily from the near-stream portion of the forest. Buffer widths of 30 m to 60 m along small (0.5-4 m) streams in New Brunswick limited post-harvest average stream temperature increases to 0.3-0.7 °C (Bourque and Pomeroy 2001). Buffer widths of 17 m to 52 m resulted in similar temperatures in small (2-4 m) streams in western Washington, whereas streamside clearcutting increased water temperatures (Brosofske et al. 1997). Similarly, partial harvest within 20-30 m riparian buffers which greatly reduced canopy closure along small (0.6-3.2 m) streams in northern B.C. resulted in temperature increases of 4-6°C five years after harvest (Macdonald et al. 2003). For undisturbed small streams, the crowns of nearbank trees may overhang the stream surface to provide very high levels of shading (Gomi et al. 2006), and understory vegetation may provide considerable cover (Kibler and Skaugset 2007). Angular canopy density measured for western Oregon streams (widths not stated) increased rapidly with riparian buffer width to asymptotes of about 80% at buffer widths near 15 m and near 30 m for old-growth forests in two separate studies (Brazier and Brown 1973, Steinblums et al. 1984), which again indicate that nearbank vegetation provides much of the potential shading. On larger streams (1-18 m), however, buffer widths less than 10 m did not prevent increases in stream temperature (median change about +1°C), but buffer widths in the ranges 10-30 m or 30-50 m both resulted in temperatures that were indistinguishable from unlogged controls (Davies and Nelson 1994).

Because riparian vegetation alters the heat budget of a stream segment, its effect is most aptly quantified in energy units as the reduction in net heat input relative to a non-vegetated state. Riparian shading would then be defined as the percent reduction in solar heat flux or irradiance to the stream segment relative to an unobstructed site (Davies-Colley and Payne 1998, Cristea and Janisch 2007).

Operational definitions of shading, however, often consider only the blockage of direct beam solar radiation (e.g., Anonymous 2000) measured or calculated at the stream surface as the percent of the sky (i.e., cover or canopy closure) or sky along the sun's path (i.e., angular canopy density) that is obstructed by vegetation and topography (Beschta et al. 1987, Davies-Colley and Payne 1998, Teti and Pike 2005) rather than as change in net energy inputs. Diffuse solar radiation and the transmission of visible and non-visible solar radiation through vegetation (Sridhar et al. 2004, Davies-Colley and Rutherford 2005) are often ignored when blockage is used as a measure of riparian shading. Thus, blockage measures such as canopy cover are not equivalent in effect to riparian shading, although they covary strongly (Kelley and Krueger 2005). However, in the common situation where the potential direct solar heat flux dominates the stream heat budget, blockage is an appropriate measure of the shading effect of riparian vegetation because changes in heat inputs will be roughly proportional to changes in blockage (Beschta et al. 1987). Blockage (measured as angular canopy density) was 75-90% along streams in undisturbed old-growth forests in western Oregon and California (Beschta et al. 1987).

The location and extent of the riparian vegetation needed to maintain normal shade regimes at a site along a stream can be determined from site geometry and vegetation characteristics by using the spatial relationships between the sun's path and the stream surface. Numerous models, of varying degrees of complexity, have been developed to estimate blockage or shading from riparian vegetation along streams (e.g., Quigley 1981, Rutherford et al. 1997, Chen et al. 1998, Bartholow 2002, Welty et al. 2002, Sridhar et al. 2004). The intensity of solar radiation received at a site depends on latitude and varies seasonally. It also varies considerably over the day, with about 60% of the unobstructed daily total being received within the six hours centred on the solar noon. Increases in stream temperature from solar heating are potentially greatest near solar noon in summer when the sun is near its zenith, solar radiation inputs are high, and stream flows are low. In this situation, the solar declination and solar zenith at a site are fixed, and stream shading is determined by the orientation of the stream with respect to the sun, tree height, distance from a tree to the stream, stream width, and vegetation density (Quigley 1981). Simple geometry then suggests that relatively low vegetation near the bank edge can provide high levels of surface shade to small streams, that tall vegetation further from the channel provides shade for wider channels, and that the proportion of the stream shaded at a given tree height and distance declines as stream width increases.

The "zone of sensitivity" (ZOS) determination for riparian shading in the RAR assessment attempts to maintain natural shading regimes over the period of stand development while utilizing a technically and administratively simple procedure that recognizes the potential influences of site potential vegetation height and stream dimensions. Site potential vegetation is used to ensure that ZOS widths incorporate

vegetation that will influence shade in the future, although it may have little current effect. This management objective rules out field measurements of current shade levels as a means of specifying a ZOS, and requires some mechanism to predict a future ZOS. While sophisticated predictors of future conditions are available (e.g., Welty et al. 2002), such data-intensive tools are impractical for sites at the spatial scale of individual urban lots. Consequently, the RAR specifies ZOS widths, similar to the fixed-width buffers used for forest management in many jurisdictions (Lee et al. 2004).

The ZOS treats the effect of vegetation height on shading by considering three height classes, based on the site potential vegetation. It assumes a site potential vegetation of “low cover” (height ≤ 1 m) will make an insignificant contribution to shading. Shrub (height ≤ 5 m) and tree vegetation height classes will contribute shade in most circumstances, but shrubs, being shorter, will contribute shade from shorter distances than trees and will be less effective on large streams. Because the summer solstice noon solar zenith for B.C. sites is about 26-36°, the shadow length cast by vegetation will be roughly 0.5 to 0.7 times the height of the vegetation at the time of maximum solar irradiance. Thus, shrubs within about 2.5 to 3.5 m of the bank and trees within about 3 m to 35 m (depending on species) can contribute to shading. These distances from which vegetation of given height can project a shadow onto the stream surface vary with stream orientation, however, and are greatest for streams oriented in an east-west direction (banks on the south and north sides). Thus ZOS widths of 2-3 m for shrubs and 3 m to 35 m for trees will encompass all vegetation capable of shading the stream surface at peak solar irradiance, regardless of stream orientation. If the height of the current vegetation is less than that of the site potential vegetation, such ZOS widths will almost certainly encompass all shade contributing vegetation. Vegetation at the bank edge will, of course, cast its entire shadow directly upon the stream surface and will fully shade a stream width up to about half the vegetation height. The proportion of the total stream surface that is shaded and the attenuation of light at the shaded surface will also depend on plant density and foliage characteristics which determine what proportion of the incident light intersects vegetation and the total thickness of vegetation through which the beam passes.

At times other than solar noon and at seasons other than the summer solstice, the solar zenith will be larger, vegetation further from the stream will intercept light that would otherwise strike the stream, and the ZOS widths required to encompass all vegetation capable of shading the stream surface will increase beyond those defined by conditions at noon at summer solstice. Maximum stream water temperatures in B.C. occur during August (Moore 2006) when stream flows have declined to summer lows. In mid-August, the solar zenith is about 35-46° at noon and shadow lengths are 0.7 to 1 times vegetation height. The effect of this additional vegetation width is further attenuating direct beam solar radiation flux at the

stream surface may be small, for several reasons. First, incident radiation intensity at the canopy is reduced, both seasonally (to about 0.87- 0.91 of summer solstice noon values) and diurnally. Second, the increase in the beam path length through the foliage (which determines attenuation) is considerably less than the increase in the maximum shadow length for trees of a given height. The 40-46% increase in shadow length at mid-August noon compared to summer solstice produces only a 10-16% increase in beam path length. Third, attenuation coefficients for solar radiation traversing the foliage may often be large enough that the absolute change in radiation flux at the stream from the increased path length is small. The attenuation coefficient for a given vegetation type will depend on foliage density, but the light attenuation observed through the canopies of undisturbed conifer forests is often 95-98% (Lieffers et al. 1999). The increase in beam path discussed above would then decrease the transmittance of the incident radiation by about an additional 1-3%. Unless the average foliage density within the riparian zone is very low, it is likely that ZOS widths estimated from the summer solar zenith will account for most of the reduction in direct beam solar radiation at the stream surface at all times.

The RAR applies maximum ZOS width of 30 m for treed site potential vegetations because angular canopy density measurements for forests similar to B.C. forests asymptote in the range of 30 m to 35 m (Beschta et al. 1987), and because few B.C. forests have average heights greater than 60 m that would contribute shade from a further distance at the time of peak solar radiation intensity. The maximum ZOS width for shrubs is set at 5 m because shrubs less than 5 m height similarly will not contribute shade from a further distance. ZOS widths for both shrubs and trees are scaled to stream width to increase the average foliage density, i.e., to improve shade “quality”, as the proportion of the stream surface potentially shaded declines with increasing stream width. In situations where shade is patchy, the increased ZOS width will also increase the total stream surface that is shaded, relative to a narrower buffer. The scalars are untested, however, and must be monitored to assess their effectiveness in limiting temperature change. The RAR regulations currently do not explicitly consider effects associated with the length of stream which is subject to disturbance. ZOS widths for lakes and wetlands are set at maximum values (5 m for shrubs and 30 m for treed sites) because 30 m forested buffers maintained littoral temperature similar to those of undisturbed sites in experimental logging studies around boreal lakes (Steedman et al. 2001) whereas maximum temperatures increased slightly at clearcut shorelines. There are no empirical data with which to assess the effectiveness of narrower shoreline buffers in maintaining normal littoral temperatures.

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Effects of riparian vegetation on bank stability

The presence and nature of riparian vegetation can greatly influence bank stability and channel form along alluvial streams (Bennett and Simon 2004, Eaton 2006) by changing local rates of sediment accretion and erosion (Beeson and Doyle 1995, Allmendinger et al. 2005). These changes arise both from the effects of vegetation on stream hydraulics (e.g., by reducing and re-directing flow velocities along banks, Thorne and Furbish 1995) and from its alteration of the mechanical and chemical properties of streambank soils (Wynn 2006). Effects vary among different vegetation types (e.g., Millar 2000, Micheli and Kirchner 2002, Simon and Collison 2002, Wynn 2004, Allmendinger et al. 2005, Simon et al. 2006, Wynn and Mostaghimi 2006a), and the removal or conversion of riparian vegetation (e.g., from forest to grass) can induce changes in the width, depth, and rate of lateral migration of the channel (Allmendinger et al. 2005, Beschta and Ripple 2006).

Streambank erosion occurs through several processes (mass wasting, fluvial entrainment, subaerial processes (see below)). Riparian vegetation reduces bank erosion by all three processes. However, the relative effects of different erosion processes and their interactions with riparian vegetation may vary systematically with position within a stream network (Lawler 1995, McKenney et al. 1995, Abernethy and Rutherford 1998, Lawler et al. 1999). Stream networks in mountainous terrain show a systematic downstream progression in channel size and in channel morphology, defined by specific ranges of channel slope, shear stress, grain size, and relative roughness (Montgomery and Buffington 1997), which reflects systematic changes in stream power (Lawler 1995, Knighton 1999) and the tractive forces acting to move bed and bank sediments as the catchment area increases. Rates of bank erosion in rivers with alluvial floodplains also increase with increasing stream power (Nanson and Hickin 1986, Larsen et al. 2006) and decrease with increasing sediment size (Nanson and Hickin 1986), presumably because erosion rates and channel morphology respond to the same forces. Thus, erosion processes and rates in alluvial streams should vary with channel type and size. Abernethy and Rutherford (1998, p.55) argue that “bank erosion in upper reaches is dominated by subaerial preparation, in mid-basin reaches by fluvial entrainment, and in the lower reaches by mass failure”. This pattern of process dominance arises, in part, from the scale-dependence of the interactions between riparian vegetation and bank erosion processes (Lawler 1995, Anderson et al. 2004). For example, susceptibility to mass wasting is influenced by the rooting depth of riparian vegetation relative to bank height, which will vary systematically with stream size. More generally, the effect of riparian vegetation on bank stability and the processes that are involved are expected to vary with the relative sizes of channel dimensions, sediments, and vegetation.

Riparian vegetation may greatly reduce mass wasting through the mechanical reinforcement of streambank soils by root networks, which increases resistance to shearing (Abernethy and Rutherford 2000, Micheli and Kirchner 2002, Simon and Collison 2002, Wynn and Mostaghimi 2006b; but see Van De Wiel and Darby 2007 [for an opposing view](#)) and may also buttress soil [from movement by point anchoring](#) (Coppin and Richards 1990). The effect of riparian vegetation on bank stability will depend on the density, size, depth, and lateral distributions of the roots (Abernethy and Rutherford 2001, Simon and Collison 2002 Simon et al. 2006, Van De Wiel and Darby 2007), the mechanical properties of the soil matrix (Wynn and Mostaghimi 2006b, Pollen 2007), and bank height. In cohesionless soils such as sands and gravels, root networks can increase measured soil shear strength considerably (Micheli and Kirchner 2002). Stabilizing effects vary among vegetation types. Wet-adapted plants such as sedges and willows impart greater strength than xeric species (Micheli and Kirchner 2002, Simon et al. 2006), partly because of differences in root architecture. Herbaceous species and grasses usually have dense networks of small-diameter roots within the uppermost 30-50 cm of the soil (Wynn et al. 2004), whereas riparian trees have larger roots that extend to 1-2 m depth (Abernethy and Rutherford 2001, Simon and Collison 2002). Root density declines with depth (Eis 1974, 1987, Jackson et al 1996). Consequently, root cohesion declines rapidly with depth (Abernethy and Rutherford 2001, Eason and Yarbrough 2002, Simon and Collison 2002). The effectiveness of different vegetation types in reducing mass wasting will thus vary with channel dimensions because erosion at the bank toe below the rooting depth of the vegetation can lead to failure from over-steepening or cantilever instability (Abernethy and Rutherford 2000, Micheli and Kirchner 2002, Eaton 2006). Root density also declines rapidly with lateral distance from the plant (Eis 1987, Abernethy and Rutherford 2001). Consequently, the stabilizing effect of riparian vegetation varies with its lateral distance from the bank face (Abernethy and Rutherford 2000, Van De Wiel and Darby 2007). Trees as far as 10-15 m from the bank can increase bank stability by 26-43% (Abernethy and Rutherford 2000). Reinforcement maxima may occur at the bank face and at the intersection of the failure plane with the floodplain surface, roughly 5 m away as modelled by Abernethy and Rutherford (2000) and Van De Wiel and Darby (2007). Failed blocks may temporarily resist entrainment and armour the bank face if root networks maintain sufficient soil cohesion (Wood et al. 2001, Micheli and Kirchner 2002).

Riparian vegetation may also influence the mechanical properties of bank soils and the strength of root reinforcement by altering soil moisture content via altered canopy interception, infiltration, and transpiration. Soil moisture content influences shear strength by altering pore-water pressure and matric suction (Fredlund 1987); lower moisture content can greatly increase soil shear strength in some cases (Simon and Collison 2002), although the effect varies with soil type (Wynn and Mostaghimi 2006b). Root reinforcement varies with soil moisture content because the shear strength of the soil matrix determines

whether roots fail by breaking or by pullout (Pollen 2007). Canopy interception by vegetation lowers soil moisture by reducing the quantity of rainfall available to infiltrate the soil and altering its spatial distribution (Keim et al. 2005). Interception varies with rainfall intensity and duration and with those vegetation characteristics (e.g., leaf area index, height) which determine the surface area that intercepts and stores rainfall or which influence evaporation from the wet surface. Although interception loss may vary considerably with site characteristics (Crockford and Richardson 2000, Pypker et al. 2005), it averages about 30% in forested areas and about 15% in grassland (Kelliher et al. 1993). Transpiration by plants also reduces soil moisture. At a given annual rainfall, catchment-scale annual transpiration by forests is considerably greater than that by grasses, especially in wetter climates (Zhang et al. 2001), although there can also be large differences in transpiration rates among species within a vegetation type (e.g., Kaufmann 1985). Differences in root distributions between vegetation types can also influence where within the soil mass transpiration effects are expressed (McLaren et al. 2004). Soil moisture content will, of course, reflect the net effect of interception, infiltration, and transpiration on precipitation inputs. Conversion of small patches of old-growth Douglas-fir forest to shrub (vine maple) resulted in higher soil moisture (Gray et al. 2002), consistent with the above general patterns.

One of the principal mechanisms by which vegetation stabilize stream banks is by reducing the erosion of particles from the submerged bank face. The fluvial entrainment of sediment depends on the near-bed stresses that result from the drag, lift, specific gravity, and cohesion forces acting on a particle (Lu et al. 2005). Bank scour, the entrainment of sediment particles from the submerged bank face, will vary with the near-bank stream velocity profile that determines the flow-induced drag and lift forces acting at the sediment surface. Particle size and any cohesive forces between particles will also influence the probability of entrainment. Riparian vegetation in contact with the stream flow will reduce the entrainment of sediment from the bank face by increasing bank roughness to reduce near-bank and bank-face water velocity (Thorne and Furbish 1995, Järvelä 2002, Wilkerson 2007) and to alter turbulence intensity (Nepf 1999). Roots and boles protruding through the bank face or vegetation growing within the active channel may also re-direct flows (Thorne and Furbish 1995, Bennett et al. 2002) to alter scour at the bank toe. They may also accumulate organic matter in transport, to further armour the bank face. During periods of overbank flow, riparian vegetation will increase flow resistance (Anderson 2006) and reduce water velocities in the near-bank region (McBride et al. 2007). However, riparian vegetation may also increase turbulence (McBride et al. 2007) and apparent shear stress (Thornton et al. 2000) at the interface between the floodplain and the channel, potentially to increase bank erosion during large overbank floods (McBride et al. 2007). The characteristics of submerged riparian vegetation that influence fluvial entrainment are those which influence its contribution to total flow resistance, such as: areal density, root

or stem diameter, submerged length, and stiffness (Stone and Shen 2002, Freeman et al. 2000, Wu et al. 2006). Its effect on near-bed stresses may vary with channel characteristics such as sediment particle size, bed planform, depth of flow, and slope.

Riparian vegetation alters other soil-surface processes that detach soil particles from the bank face. “Subaerial processes”, such as frost heave, desiccation cracking, and rainsplash, are physical-chemical processes that detach soil particles to deliver sediment directly to the channel from the bank face. They also increase the effectiveness of fluvial entrainment and mass wasting by reducing soil strength (Wynn 2006). Subaerial processes can be a major component of bank erosion in fine-grained soils (Couper and Maddock 2001), particularly along headwater streams (Abernethy and Rutherford 1998, Wynn and Mostaghami 2006a). Riparian vegetation alters subaerial erosion rates by moderating fluctuations in soil temperature and moisture content, e.g., the timing and frequency of freeze-thaw cycles or the spatial distribution of desiccation (Stott 1997, Wynn and Mostaghami 2006a). Vegetation effects on subaerial erosion vary with vegetation type and density, as well as with stream characteristics such as bank height and soil type (Wynn and Mostaghami 2006a).

The qualitative effects of riparian vegetation on bank stability are clear: vegetated riparian areas will generally increase geotechnical stability and decrease fluvial and subaerial erosion along stream banks. Higher vegetation density and/or higher root density will generally result in greater bank stability by increasing bank shear strength. It will also reduce flow velocities at the bank face by increasing hydraulic roughness, as will larger root or stem dimensions relative to bank sediment dimensions. Because the erosive forces acting on the bank to induce fluvial entrainment of bank sediments increase with increasing water velocity while resisting forces increase with bank particle size, the role of riparian vegetation in stabilizing stream banks will vary with channel type and size. Observed erosion rates in low-gradient alluvial streams generally increase with channel width (Nanson and Hickin 1986). The effect of riparian vegetation on bank stability will be relatively high in riffle-pool type alluvial channels whose bank materials are small relative to channel and vegetation dimensions and are easily moved by flowing waters, especially for non-cohesive sands and gravels. The dense but shallow root mats of herbaceous vegetation can stabilize the banks along small riffle-pool channels with low stream power, but their relative effect will diminish as stream size and power increases. Riparian trees can stabilize the banks of larger riffle-pool channels until bank height exceeds rooting depth, at which point undercutting below the root mat may cause cantilever failures. As the relative roughness of bed and bank sediments increases in alluvial cascade-pool and step-pool channels, the relative contribution of vegetation to soil cohesion and bank stability may diminish, although roots and boles that are large relative to bank sediment size will still

reduce fluvial entrainment at the bank face. The larger and deeper roots of trees may contribute more to bank stability along such stream types than those of shrubs or low cover. The role of riparian vegetation in maintaining bank stability will be further reduced in small colluvial headwater channels whose (generally) coarse bank materials may often exceed channel competence.

The quantitative effects of riparian vegetation on bank stability are much more difficult to specify. Vegetation at or near the bank face will directly influence both fluvial entrainment and bank stability. Vegetation effects on bank stability can extend about 15 m laterally and 1 to 2 m vertically for some tree species, but will be much less for herbaceous plants or shrubs and for many common B.C. tree species (e.g., Sakals and Sidle 2004). Because the unvegetated or grassed land upslope of riparian buffers in urban areas will often have much lower stability, treed ZOS widths should likely exceed 15 m to accommodate lateral channel movement and to reduce the risk that channels migrate through the buffer, if such movement is evident or expected. Beechie et al. (2006) noted that low gradient (slope < 4%) alluvial channels in Puget Sound and the Olympic Peninsula did not migrate laterally through adjacent forest until their bankfull widths exceeded 15 to 20 m, at which size their bank heights exceeded the rooting depths (0.5 to 0.75 m) of mature conifers, and mass failure by bank undercutting commenced. This observation suggests a threshold channel size below which mature conifers are effective at stabilizing stream banks in coastal areas. The observation by Hall et al. (2007) that stream segments (slope < 20%) in the Yakima River basin with widths less than 8 m did not migrate laterally further suggests that conifer forests are effective in stabilizing banks below a threshold channel size in the range of 10-20 m. Erosion rates in large, low-gradient, meandering alluvial streams increase with channel width, again through basal undercutting (Nanson and Hickin 1986), which suggests that ZOS widths for bank stability may become ineffective above a threshold channel size if sediments at the bank toe can be fluvially entrained. Rooting depth relative to channel depth may suggest whether lateral migration by bank instability is likely to be influenced by riparian vegetation.

Confinement ratios (floodplain width/channel width) provide direct information on the extent of long-term lateral movement of the channel of alluvial streams. Montgomery and Buffington (1997) state that confinement varies systematically along drainage networks, generally decreasing in an upstream direction as part of a suite of covarying traits (slope, discharge, relative roughness) that define channel types. O'Connor and Watson (1998) also note that valleys generally narrow with increasing slope and that confinement is rarely > 2 when channel slope is > 8%. They state that "significant channel migration occurs infrequently where channel slope > 6%". Hall et al. (2007) found that stream segments with confinement ratios < 3.8 did not migrate laterally. These data suggest that the extent of lateral adjustment

expected in steep-gradient channels (step-pool, cascade-pool types) is roughly 2-4 times the channel width. Thus, ZOS widths whose two-bank extent is 2-4 channel widths will likely encompass most channel movement in steep-gradient channel types.

The RAR adopts a highly simplified approach to specifying riparian ZOS widths for bank stability which is based on channel type (i.e., slope and relative roughness), channel size, and site potential riparian vegetation type. The minimum ZOS width at all treed channels (10 m) is roughly the average maximum radius of root penetration for hemlock, red cedar, and Douglas-fir (Eis 1974); thus a tree at the outer boundary may contribute to bank strength near the channel. ZOS widths for all channel types increase with stream size (to some maximum ZOS width) in order to accommodate increasing shear stresses and possible lateral migration of the channel. Single-bank ZOS widths for treed steep-gradient step-pool and cascade-pool channel types increase as multiples of channel width ($1\times$ or $2\times$) to encompass most of the expected channel migration zones (i.e., confinement ratios of 3-5) for small channels. The ZOS width for treed riffle-pool channels increases as $3\times$ the channel width and will encompass expected movement for small streams with confinement ≤ 7 . The maximum ZOS widths for treed riffle-pool channels (30 m) occurs more-or-less at the channel width (10 m) above which lateral migration via mass failure is expected to become important and vegetation less influential in determining bank stability. The RAR does not further increase ZOS widths because more extensive channel movement along small urban streams is likely to be constrained by dyking to protect property and infrastructure. Maximum ZOS widths for a given channel type are reduced for shrub and low cover site potential vegetations whose laterally less-extensive and shallower root systems (Schenk and Jackson 2002) can reinforce the basal sediments of the banks of small streams but which will be undercut at smaller channel widths (i.e., bank heights) than deeper-rooted trees would be.

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Large woody debris inputs to streams

The importance of large woody debris to stream habitats

Riparian forests supply wood to adjacent stream channels (McDade et al. 1990, Benda et al. 2002). Such riparian-derived wood may interact with stream flow to alter the hydraulics, sediment dynamics, geomorphology (Keller and Swanson 1979, Gurnell et al. 2002), and biological productivity (Smock et al. 1989, Wallace et al. 1995) of the receiving stream reach. The fate and effects of riparian-derived wood subsequent to its entry to the stream are strongly influenced both by characteristics of the piece such as its diameter, length, orientation, density, and the presence of a rootwad and by those of the channel such as depth of flow, width, gradient, surface roughness, and the presence of obstructions (Nakamura and Swanson 1994, Braudrick and Grant 2000, Haga et al. 2002, Bocchiola et al. 2006a). Wood whose dimensions are large relative to channel dimensions will tend to remain near its entry location (Hilderbrand et al. 1998) and may exert a significant influence on the physical processes that shape the channel. Smaller pieces of wood may remain temporarily near their entry points, but will be transported downstream during high discharge (Haga et al. 2002) unless they become incorporated into stable debris jams initiated by a large “key” piece of wood (Abbe and Montgomery 2003).

“Large” woody debris is often defined operationally as the minimum-sized piece that will induce geomorphic change. A stable piece of wood will locally modify the water surface profile (Bocchiola et al. 2006b) and flow velocities (Abbe and Montgomery 1996), altering patterns of energy dissipation at a site. In alluvial channels, the flow changes may induce scour adjacent or beneath the piece (Keller and Swanson 1979, Abbe and Montgomery 1996, Buffington et al. 2002) and sediment deposition in low velocity zones upstream (Megahan 1982, Montgomery et al. 1996) and/or downstream (Keller and Swanson 1979) of the piece. The pattern of scour and deposition induced by stable wood within an alluvial channel increases the frequency of pools (Montgomery et al. 1995) and the total area or volume comprised by pools (Bisson et al 1987, Buffington et al. 2002). Heterogeneity of sediment texture (Buffington and Montgomery 1999) and of channel gradient (Faustini and Jones 2003) also increases.

Large woody debris will also alter the biological productivity of a stream reach. The alterations to stream hydraulics induced by woody debris retard the downstream transport of water, solutes, and materials in suspension (Gurnell et al. 2002). Stable wood within stream channels traps particulate organic matter (Bilby and Likens 1980, Smock et al. 1989) and modifies the storage and release of nutrients (Bilby 1981, Valett et al. 2002), potentially to increase the production of benthic invertebrates (Wallace et al. 1995). Instream wood also increases the surface available as a substrate for heterotrophic and autotrophic micro-organisms (Sinsabaugh et al. 1991), serves directly as a carbon source for xylophagous organisms

(Anderson et al. 1978, Collier et al. 2000), and produces fine particulate organic matter (Ward and Aumen 1986). It may also serve as protective overhead cover for fishes, independently of its geomorphic function (Shirvell 1990). In general, the diversification of habitats induced by stable wood within a stream channel improves conditions for stream-rearing salmonid fishes (House and Boehne 1986, Fausch and Northcote 1992, Roni and Quinn 2001). Maintaining normal quantities of wood in stream channels is therefore an important goal for the management of riparian areas.

Wood enters a stream reach from the adjacent riparian forest via several processes, including: bank erosion, overbank flooding, tree fall from natural mortality or windthrow, and landslides (Swanson and Lienkaemper 1978, Keller and Swanson 1979, Murphy and Koski 1989, Palik et al. 1998, Benda et al. 2002, Acker et al. 2003, Benda et al. 2003). It may also enter from upstream reaches by fluvial transport (Kreutzweiser et al. 2005) or by debris torrents (May and Gresswell 2003). Wood is removed from the channel by burial and by fluvial transport as pieces decay and/or fragment to a size that can be moved by prevailing flows. The quantity of wood found within a stream reach at any time thus will reflect a shifting balance between input and removal processes. Where inputs are greatly reduced for long periods, as when riparian vegetation is removed and the regenerating vegetation is still too small to produce stable wood, instream quantities of stable, “functional” wood will decline (Murphy and Koski 1989, Bilby and Ward 1991) and stream channels may evolve towards a simpler geomorphic structure (Bilby and Ward 1991) that may have a reduced ability to produce fish-food organisms (Fausch and Northcote 1992) or fish. Reduced wood input from cleared riparian areas may persist for many decades (Murphy and Koski 1989).

To avoid these undesirable effects, many resource management agencies have adopted regulations intended to preserve riparian forest and maintain normal inputs of wood. Commonly the protective regulations take the form of mandatory, prescribed, undisturbed or minimally disturbed riparian setbacks (“buffers”) adjacent to the stream channel (Lee et al. 2004). Because of the value of the merchantable timber left within riparian buffers, and in urban areas because of the value of the cleared land itself, the width of the riparian buffer that is necessary to maintain normal inputs of large woody debris to a stream has frequently been a contentious issue. In urban settings where, once cleared, riparian areas will remain untreed, ensuring that sufficient riparian forest is retained in buffers to assure the adequate long-term provision of LWD is particularly important.

Factors that influence source distances for the delivery of functional large woody debris to streams

The lateral distance over which wood is delivered to a stream channel from the adjoining forest depends on characteristics of the vegetation and on the dominant delivery mechanisms operating at a site.

Tree density, size, and mortality rate, and lateral zonation in these (e.g., transitions between true riparian and upland plant communities along soil-moisture gradients), will influence the quantity, piece size, and origin of wood entering the active stream channel. The physical process by which wood enters the channel influences the lateral distance from which it is recruited and the resulting piece size. The processes that commonly move wood directly from adjacent areas into the stream channel (bank erosion, tree fall or windthrow, landslides) operate over different lateral distances (Benda et al. 2002, May and Gresswell 2003). Bank erosion will only deliver trees at the current channel edge (Murphy and Koski 1989, Benda et al. 2002), although unconfined alluvial channels wider than about 15 m have sufficient stream power to migrate laterally across forested floodplains (Beechie et al. 2006) and so may obtain LWD over greater lateral distances than smaller riffle-pool channels. Severe flooding can recruit wood from the near channel portion of the active floodplain (Palik et al. 1996, Acker et al. 2003), but on unconstrained reaches much of the wood may actually be delivered through bank erosion. Dead fall will deliver LWD from distances up to about one tree height from the channel (Van Sickle and Gregory 1990). Similarly, windthrow will normally deliver wood only within one tree height of the channel (Grizzel et al. 2000, May and Gresswell 2003). Landslides may introduce wood from longer distances (Benda et al. 2002, May and Gresswell 2003, Reeves et al. 2003) if slopes are connected to the channel, but will normally occur only on steep sideslopes with low-cohesion overburden (Swanston and Howes 1991).

The relative importance of the different mechanisms by which wood is delivered to an adjacent stream channel varies with channel characteristics (Lienkaemper and Swanson 1987, Murphy and Koski 1989, Martin and Benda 2001, Reeves et al. 2003). The systematic changes in channel size, gradient, substrate, and hillslope coupling along drainage networks which define channel types reflect changes in the processes that deliver and transport sediments (Montgomery and Buffington 1997). LWD inputs respond to the same processes, e.g., Martin and Benda (2001) found that the recruitment of LWD from bank erosion increased systematically with drainage area in an Alaskan watershed. Differences in LWD input mechanisms among channel types can be thought of conceptually as systematic variation in rates of landslides and bank erosion along the drainage network, superimposed on a background rate of tree fall that is less strongly associated with channel form. In general, we expect landslides and tree mortality to be the dominant LWD inputs in steep-gradient, confined, bedrock or coarse-textured channels with steep valley sideslopes while bank erosion will increase in importance in lower-gradient, unconfined, alluvial channels with broader valley flats. As a result, the lateral distances over which LWD originates will vary with channel type and size (Johnston et al. 2007). The local influences of topography, connectivity, and tree mortality may modify these expectations, however (Benda et al. 2002, Reeves et al. 2003).

Variation in LWD functioning among channel types

Structural elements such as clasts, individual LWD pieces, or LWD accumulations whose dimensions are large relative to channel dimensions often exert a controlling influence on the morphology of alluvial channels (Church 1992, Gurnell et al. 2002) by altering local sediment transport and deposition. The importance of LWD pieces as channel-shaping agents will depend on their size and abundance relative to other large roughness elements within the channel. It will also vary with factors such as channel gradient, width-to-depth ratio, grain size relative to flow depth, and the size and rate of supply of bed material (Buffington et al. 2002). Steep-gradient channels in mountainous terrain (i.e., step-pool and cascade-pool channel forms) generally have a capacity to transport sediment that is high relative to sediment supply (Montgomery and Buffington 1997). Large clasts or stone lines that are immobile except at extreme flows form important components of the bed material of such channels and are often the structural elements that determine channel morphology. LWD may have only a minor influence on channel morphology in steep-gradient channels (Montgomery et al. 1995, Anonymous 1996, Berg et al. 1998, Warren and Kraft 2003), although channel-spanning pieces may form steps that induce “forced” riffle-pool morphology if sediment inputs are sufficient (Montgomery et al. 1996, Faustini and Jones 2003), and channel width may increase with the frequency of flow obstructions (Jackson and Sturm 2002). In small, constrained channels much of the LWD may lie above the active channel and have no immediate geomorphic effect (Halwas and Church 2002, Chen et al. 2006). These steep-gradient channels typically have erosion-resistant banks, high depth-to-width ratios, and high connectivity to hillslopes (Montgomery and Buffington 1997). Consequently, LWD inputs are dominated by landslides and tree fall (Lienkaemper and Swanson 1987, May and Gresswell 2003, Montgomery et al. 2003). The steep sideslopes and high connectivity increase LWD loadings per unit stream length by increasing the recruitment area for woody debris per unit horizontal distance (Jackson and Sturm 2002).

LWD pieces that are stable in low-gradient, riffle-pool channels are considerably larger than the bed material, and can greatly alter channel morphology by causing localized scour and deposition. Mean pool spacing (Montgomery et al. 1995), pool area (Hogan 1986, Buffington et al. 2002), variability in channel width and depth (Hogan 1986), and sediment storage (Bilby and Ward 1989) all increase with increasing LWD loading in low-gradient channels. Low-gradient riffle-pool channels are characterized by lower depth-to-width ratios, smaller and more-easily mobilized bed materials, and well-developed alluvial floodplains (Montgomery and Buffington 1997). Banks of fluvially deposited materials and reduced connectivity with hillslopes result in bank erosion and undercutting being relatively more important as LWD delivery mechanisms in these channels.

The stability and geomorphic functioning of a LWD piece will be influenced by both channel type and channel size. At a given channel width, the steeper gradient and greater depth-to-width ratio of a step-pool or cascade-pool channel compared to a riffle-pool channel generally result in higher stream power, greater submergence, higher obstruction ratio, and higher net forces on a LWD piece of a given size. Thus, the minimum size of a stable LWD piece is expected to be larger in the steeper gradient channels than in a riffle-pool channel of the same width. The mean size of functional LWD also increases with channel width (Likens and Bilby 1982, Bilby and Ward 1989, Chen et al. 2006), presumably because increased stream power removes small wood. Other things being equal, a smaller proportion of woody debris inputs will be geomorphically functional at a steep gradient channel than at a similar-size riffle-pool channel, and the steeper channel will export more small wood to downstream reaches.

In general, the effect of LWD on channel structure is expected to be greatest in small to medium-sized, low-gradient stream reaches (Anonymous 1996). Although pool-formation in small steep-gradient streams can be influenced by LWD (Montgomery et al. 1995), and the rate of change of pool spacing with increasing LWD loading is greater in moderate gradient streams than in low-gradient streams (Beechie and Sibley 1997), pool numbers and area at a given loading still are greater in low gradient streams (Beechie and Sibley 1997, Buffington et al. 2002). The upslope delivery mechanisms (debris flows, landslides) that are relatively more common at constrained, steep-slope reaches introduce woody debris which is considerably smaller than that originating from streamside mechanisms (Reeves et al. 2003). A higher incidence of bank erosion at riffle-pool channels should result in a greater frequency of large piece sizes (e.g., whole trees with root-wads) entering than at step-pool and cascade-pool channels for the same vegetation density and size. Because the geomorphic impact of LWD (pool size, sediment storage) increases with piece size (Bilby and Ward 1989) and LWD abundance per unit stream length is highest in low-gradient, unconfined channels (Martin 2001, Fox and Bolton 2007), low-gradient alluvial channels are expected to be strongly influenced by LWD inputs. Moreover, large wood pieces that are well-anchored (e.g., by root-wads) can act as key pieces (Abbe and Montgomery 2003) that accumulate smaller pieces in transport to form stable logjams which may alter channel morphology over very long distances in low-gradient channels (Hogan et al. 1998). Experimental additions and removals of LWD in small streams generally show that the presence of LWD alters channel structure in low-gradient streams but has little effect in mid- and high-gradient streams (Hilderbrand et al. 1997, Roni et al. 2002, Warren and Kraft 2003, Sweka and Hartman 2006); effects on invertebrates and fish are variable (Wallace et al. 1995, Hilderbrand et al. 1997, Roni et al. 2002), possibly because of statistically-weak experimental designs.

In addition to direct geomorphic effects, the presence of stable LWD within the active channel may influence other aspects of fish habitat beneficially, and its reduction or removal may alter habitat quality. LWD retards the downstream transport of sediment, particulate organic matter, water, and solutes. Stable LWD increases sediment storage in both steep- and low-gradient streams (Megahan 1982, Bilby and Ward 1989, May and Gresswell 2003). In some cases, the storage of gravels by LWD improves the quality of spawning gravels used by salmonids or creates spawning habitat in high-gradient channels where it could not otherwise exist (Montgomery et al. 1999). LWD also increases the retention of (riparian-derived) particulate organic matter such as leaves, needles, twigs, and branches (Smock et al. 1989, Ehrman and Lamberti 1991, Raikow et al. 1995, Daniels 2006) which serve both as carbon and nitrogen sources and as physical habitat for the biota of heterotrophic streams (Richardson 1991, 1992). Removal of LWD accumulations can greatly increase the export of organic matter, both solutes and particulates (Bilby and Liken 1980, Bilby 1981). Retention of organic matter by LWD may be especially important in small streams where inputs of coarse particulate organic matter per unit stream area are highest (Connors and Naiman 1984). The heterogeneity in micro-habitat features (substrate, current velocity) and alterations to organic matter storage induced by LWD can increase the diversity and alter the productivity of benthic macro-invertebrates in small low-gradient (Smock et al. 1989) and medium-gradient (Wallace et al. 1995) streams.

Conceptual framework for the “zone of sensitivity” determinations for LWD delivery

The widths of the “zone of sensitivity” for LWD recruitment prescribed by the Riparian Area Regulations of the Fish Protection Act vary with stream channel type, bankfull width, and site-potential vegetation type (Table 1c). The matrix framework attempts to recognize the differing delivery processes and functional roles of LWD in forested streams of different channels types and sizes. The differences in the zone of sensitivity for LWD recruitment among channel types, channel widths, and site-potential vegetation types reflect our current understanding of the origins and functions of LWD in streams in urban areas. Because unstable slopes that may deliver LWD through landsliding or debris flows are delineated and included in the zone of sensitivity determination by a separate process (“measures ... necessary to protect the integrity” of the ZOS), only bank erosion and tree fall are considered as LWD input mechanisms in Table 1. The key assumptions on which Table 1 is based are that:

1. low-gradient streams whose site-potential riparian vegetation is forest would normally have a forced riffle-pool channel form which is sensitive to LWD loadings,
2. the structure of higher-gradient cascade-pool or step-pool streams is relatively less sensitive to LWD loadings because stone lines or large clasts are present and act as form elements,

3. the stability and geomorphic effect of a piece of wood increases with the size of the piece relative to channel dimensions (i.e., as piece diameter/bankfull depth or piece length/bankfull width increase),
4. for a given bankfull discharge, the minimum size of a piece of wood that can alter channel morphology increases with increasing channel gradient,
5. the size of a piece of wood entering a stream channel from the adjoining riparian area increases, on average, as the lateral distance from the bank edge to the rooting location decreases,
6. for a given channel width, bank erosion is relatively more important as a LWD delivery mechanism in low-gradient, alluvial, riffle-pool channels than in cascade-pool or step-pool channels,
7. rates of bank erosion increase with discharge (i.e., with channel size) and cause riffle-pool channels to migrate laterally within their alluvial floodplains,
8. tree fall (i.e., dead fall, windthrow, stem breakage) will deliver LWD originating within a lateral distance of about one site-potential tree height from the channel,
9. steep side-slopes and high connectivity between hillsides and stream channels, which are characteristic of steep-gradient channel types, increase the recruitment area per unit lateral distance for tree fall inputs of LWD,
10. the probability of export of wood of a given size from its point of entry to downstream reaches increases with channel size and gradient, and
11. LWD that does not directly alter the input, storage, or transport of sediment or water at a site may nevertheless be required as a nutrient source, a substrate for attachment, or cover.

A conclusion that may be drawn from these assumptions is that providing adequate inputs of LWD is particularly important in riffle-pool and forced riffle-pool channels whose physical structure responds strongly to the presence (or absence) of stable wood. Moreover, wood-dependent, structurally-complex, low-gradient riffle-pool channels are often important spawning and rearing habitat for salmonids (Montgomery et al. 1999) and sites of high species richness and fish abundance (Platts 1979, Beecher et al. 1988). LWD at low-gradient, riffle-pool channels may enter by bank erosion, overbank flooding, tree fall, or fluvial transport from upstream areas. Although the largest and geomorphically most influential pieces will originate near the current channel margin (by both erosion and tree fall), the position of the channel within its alluvial floodplain will change over time through lateral erosion. While riparian vegetation will slow erosion, the rate of erosion generally increases with channel size and low-gradient alluvial streams greater than 15-20 m width can migrate laterally through forested floodplain (Beechie et

al. 2006). In contrast, small steep-gradient channels in British Columbia often have stable channels with abundant large clasts and are less dependent on wood as a structural element (e.g., Halwas and Church 2002). While LWD may enter such streams from the undercutting of trees along the bank, most LWD will enter from tree fall (in the absence of landsliding). Steep side-slopes will increase the relative importance of tree fall inputs by increasing the effective area from which LWD is recruited to the stream by tree fall within a given lateral distance. However, steeper channel gradients and narrower, deeper channels per unit discharge will result in functional LWD being larger in step-pool and cascade-pool channels compared to riffle-pool channels of the same width. The combination of a greater slope distance per unit lateral distance and a larger piece size requirement for wood to be functional in steeper-gradient channels means that wood originating at a lateral distance which contributes the minimum size of functional LWD to a riffle-pool channel will not be functional in steeper channels of the same width. That is, the maximum lateral distance from which tree fall contributes stable wood will be reduced in steep-gradient, constrained channels with steep side-slopes.

These qualitative patterns and their underlying assumptions are supported to varying degrees by currently available empirical data, as discussed above. There are, however, only limited observational data with which to test expected patterns and to establish quantitative zones of sensitivity. Perhaps as a consequence, many discussions of LWD inputs to streams implicitly assume tree fall as the dominant LWD recruitment mechanism, and adopt one site-potential tree height as an appropriate buffer width for the provision of LWD. The widely-known data of McDade et al. (1990) and the FEMAT synthesis (see Fig. 1) both suggest that riparian buffer widths of one tree height will encompass most LWD sources. A recent review of all available published LWD source distance data (Johnston et al. 2007) indicated, however, that LWD source distances were generally less than one tree height and varied by channel type and size through a significant channel type \times channel size interaction. Tree height had a strong influence on LWD source distances, nevertheless. Ignoring channel type or size, the median source distance for 90% cumulative inputs of LWD was about 0.5 tree heights, although there was considerable variation at a given tree height. At sites where erosion was identified as the dominant process for LWD input, source distances were reduced compared to sites where tree fall was the dominant process. In general, source distances at riffle-pool channels were lower than those at steeper-gradient channel types and varied little with channel size, although the range of source distances appeared to increase at larger channels; this pattern likely reflected the importance of erosion as a LWD input process at riffle-pool channels. LWD source distances at small cascade-pool and step-poll channels were similar to those at riffle-pool channels but source distances increased with channel size at these steep-gradient channel types; possibly this indicated a greater role for tree fall and/or landslides. Although these patterns support varying ZOS widths for LWD

recruitment by channel type and size, the empirical data are very limited and come from locations somewhat dissimilar to those where the RAR will apply. Ongoing data collection within B.C. will ultimately provide data that are more suitable for delineating ZOS for LWD inputs.

The ZOS widths currently specified in the RAR (Table 1c) were elicited from B.C. researchers familiar with the role of LWD in small streams, supplemented by a review of available data from elsewhere. As suggested by the available empirical data, ZOS widths within vegetation height classes and channel types are increased with increasing channel size, to a maximum value. Particular attention is given to the provision of LWD to treed riffle-pool channels where its geomorphic function is relatively greater than in steeper-gradient channel types, and where smaller pieces can be functional at a given stream width. Chronic inputs of LWD to riffle-pool channels appear to originate mainly from erosion at near-bank riparian areas. Because bank erosion causes riffle-pool channels to migrate laterally within their active floodplains, ZOS widths to provide LWD to such channels are determined by those established to maintain bank stability. Maximum ZOS widths for riffle-pool channels (30 m) will usually exceed 0.5 tree heights for B.C. tree species, will equal one tree height in many biogeoclimatic zones, and should often encompass most tree fall inputs. Along steep-gradient channel types, high connectivity between hillslopes and channels can result in episodic landslides delivering wood to the channel from relatively long distances. However, areas of potentially unstable terrain are delineated by a separate process within the RAR, so ZOS widths for LWD inputs to cascade-pool and step-pool channels consider only erosion and tree fall as delivery mechanisms. Although tree fall might be expected to be a relatively important delivery mechanism to these channel types, professional experience and limited (unpublished) data suggest that most functional LWD enters from relatively short lateral distances. The maximum ZOS widths for cascade-pool and step-pool channels reflect this belief, which must be tested with further data collection.

The widths specified in Table 1c are based on professional opinion for several reasons. First, published LWD source distance curves generally do not separate channel types or widths. Second, published LWD source distance data were determined at geographic areas whose vegetation, climate, geology, and topography may not be representative of the regions to which the RAR will be applied. Third, there are no published LWD source distance curves for B.C. streams to guide the selection of appropriate widths. An ongoing program of data collection will ultimately lead to revised values that better represent natural patterns of LWD recruitment to streams in the geographic areas in which the RAR will be applied. It should also be noted that urban streams are managed systems: while expensive, undesirable, and sometimes ineffective, it is nevertheless possible to add or otherwise manipulate LWD characteristics in urban streams should current regulations not provide desired conditions and benefits.

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Large woody debris in lakes and wetlands

Large woody debris occurs commonly in the littoral areas of lakes (Christensen et al. 1996, Mallory et al. 2000, Francis and Schindler 2006, Marburg et al. 2006), where it can be an important component of fish habitat (Newbrey et al. 2005, Sass et al. 2006). Littoral LWD increases the structural complexity of shallow-water environments, providing visual and physical isolation that may influence predator-prey interactions among fishes (Everett and Ruiz 1993, Sass et al. 2006) to alter feeding and growth. The presence of LWD alters the spatial distribution of fishes by attracting fish (Moring et al. 1986); fish use (both abundance and taxonomic richness) increases with the structural complexity of the piece (Newbrey et al. 2005). LWD in lakes provides a physical substrate for microbial, periphyton, and insect production (Bowen et al. 1998, Smokorowski et al. 2006), as well as a spawning substrate for some fishes (Scott and Crossman 1973). Its presence may alter the dynamics and physical characteristics of sediment in the nearshore environment.

The abundance of LWD (pieces per unit shoreline) varies with riparian tree density within 10 m of the shoreline (Christensen et al. 1996, Francis and Schindler 2006) or the density of riparian LWD within 10 m of the shoreline (Marburg et al. 2006), both among and within lakes. Littoral LWD abundance declines with increasing lakeshore development, both among and within lakes (Christensen et al. 1996, Marburg et al. 2006, Francis and Schindler 2006). Removal of littoral LWD can induce large declines in some fish species (Sass et al. 2006). Because submerged wood persists for extremely long periods (Guyette and Cole 1999), adverse effects on fishes of human-induced declines in LWD abundance will likely be long-term. These observations suggest that site-level reductions in LWD inputs can produce persistent deleterious effects on fish.

There are no published studies of the recruitment of LWD to lakes. However, processes such as shoreline erosion, tree fall (from chronic or episodic tree mortality, beaver activity, wind throw, fire), and flooding of nearshore riparian areas are likely to be the sources of most natural LWD, and will recruit wood from lateral distances similar to those observed along streams. Newly-input LWD may be re-distributed by flotation, but the observed correlations between littoral LWD abundance and site-level riparian tree density suggest that re-distribution is relatively minor. The observed correlations, over a wide range of tree densities, between littoral LWD abundance and either tree density or riparian LWD density within 10 m of the lake suggest that much of the LWD may originate within the 15 m RAR zone of sensitivity.

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