A low-risk strategy for preserving riparian buffers needed to protect and restore salmonid habitat in forested watersheds of Washington state

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Summary

In this document, we provide the scientific basis for determining riparian buffer widths needed to protect streams from certain harmful effects of logging-related activities, or "forest practices" as they are referred to in Washington state law. Based on available scientific information, we recommend specific riparian widths for fish bearing and non-fish bearing streams that will provide fully functional riparian forests over the long-term and thus minimize the risk that continued degradation of riparian habitat will result in the extinction of salmonids at any time in the future.

Protecting salmonid habitat over the long-term means providing adequate protection for all areas that are or might become aquatic habitat at some point in the future. This means that floodplains, channel migration zones, and small valley floors that could be flooded by beaver dams, all need to be protected. Outside of these areas, riparian buffers are needed to ensure that current and future aquatic habitat is adequately protected and supplied with appropriate levels of material and energy inputs. We determined the size of riparian buffers needed to provide stream environments with levels of large woody debris (LWD), small woody debris (SWD), litterfall, shade and relative humidity that approximate natural conditions. We also determined the width of riparian buffers necessary to remain windfirm. We did not determine the protection necessary to ensure that sediment loading to streams returned to approximately natural levels since this requires protecting unstable slopes, areas that are often outside of what are normally referred to as riparian buffers.

To eventually have instream levels of LWD and SWD that approximate natural conditions, a buffer width of one 300 year site potential tree height ($SPTH_{300}$) is needed. In western Washington, $SPTH_{300}$ generally range from 105-250 feet, while in eastern Washington, they range from 50-250 feet. To maintain instream litterfall rates at natural levels requires buffer widths of one-half a $SPTH_{300}$, while buffers become relatively windfirm when they are wider than 75 feet. In order to provide shade to streams that approximates natural conditions, buffer widths of 250 feet are required. Likewise, 250 foot buffers are necessary to maintain relative humidity levels near the stream at natural levels.

Therefore, in order to fully protect and restore riparian habitat upon which salmonids depend, interim buffer widths of 250 feet are proposed for all perennial streams and a width equal to one full site potential tree height (50-250 feet) on all seasonal streams. These buffers are intended to ensure that riparian forests return to as close to 100% functionality over the long-term as is reasonably possible, and that the future condition of riparian forests does not contribute significantly to the loss of salmonid populations. The rationale for these buffer widths is based on the best, currently available scientific information.

These buffers are considered interim because as more data becomes available, the widths of these buffers may change. For example, on smaller perennial streams, buffers narrower than 250 feet may, in certain instances, still provide close to 100% riparian function (once they recover from their current degraded state).

While these proposed buffer widths will ultimately minimize the negative affects of riparian conditions on salmonid populations, the continued existence of salmonids in forested watersheds is also dependent on adequate protection elsewhere. In particular, forest practice activities on unstable slopes need to be minimized, and problems resulting from extensive logging road networks still need to be addressed.

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Introduction

A credible plan to protect and restore salmonid habitat on private forest lands in Washington state is long overdue. Many salmonid runs within the state are in critical condition and many stocks are already extinct (Nehlsen et al. 1991). The rate at which stocks are declining is alarming and suggests that substantial improvements in salmonid habitat protection need to occur immediately to prevent remaining stocks from becoming extinct. Existing regulations provide minimal levels of habitat protection and fail to adequately protect endangered species (Stelle 1998). To prevent the extinction of dwindling salmonid runs, changes in watershed management that protect and restore salmonid habitat in a meaningful time frame need to be adopted.

In this document, we provide the scientific basis for determining riparian buffer widths needed to protect streams from the harmful effects of logging-related activities, or "forest practices" as they are referred to in Washington state law (WFPB 1995a, WFPB 1995b). We also briefly discuss additional protection outside the riparian zone that will be needed. Based on our interpretation of available data, we recommend specific riparian buffer widths for fish bearing and non-fish bearing streams that will provide fully functional riparian forests over the long-term and thus minimize the probability that continued degradation of riparian habitat will result in the extinction of salmonids at any time in the future.

Fundamental components of a salmonid habitat conservation strategy

In forested watersheds, salmonid habitat is primarily degraded by three logging-related activities: 1) road construction, 2) logging on unstable slopes and 3) logging in riparian zones (see reviews in Meehan 1991, FEMAT 1993). As such, a salmonid habitat protection strategy needs to adequately protect unstable slopes and riparian areas, as well as ensuring that roads are constructed properly. If these three activities are properly regulated, freshwater salmonid habitat is more likely to recover over the long-term. To the extent that these activities are not properly regulated, there will be an increasing level of uncertainty as to whether salmonid populations persist. As a hedge against this uncertainty, healthy areas where entire watersheds are managed as natural reserves, need to be included as a fourth major component of a salmonid habitat protection plan (sensu FEMAT 1993). While detailed plans for all these components are important, this paper focuses on developing a scientific rationale for adequately protecting riparian habitat.

Riparian buffers are the key component of any salmonid habitat conservation strategy because they occupy the greatest amount of land area and provide the majority of the ecological goods and services required to keep salmonid habitat functional. However, strategies to protect unstable slopes, ensure that roads are correctly built and maintained, and protect watershed refugia are also quite important. Thus, the recommendations presented here are adequate for protecting salmonid habitat only if additional protection is provided outside of the riparian zone.

Functional components of riparian buffers

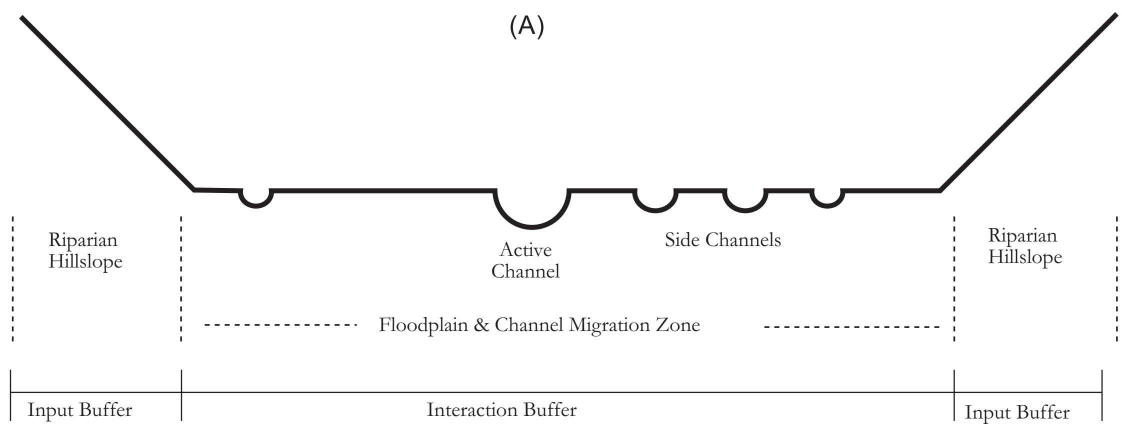
The term riparian refers to the environment adjacent to waterbodies such as streams, lakes and wetlands. Naiman et al. (1998) use the term "riparian forest" to refer to vegetation directly adjacent to rivers and streams, and includes the floodplain and any adjacent area that can contribute organic matter such as large wood debris (LWD) to the active channel or floodplain (sensu Gregory et al. 1991). We generally agree with this definition, but modify it to include contributions other than organic matter, and include areas adjacent to all waterbodies, not just streams. Thus, while we refer to the effects of riparian forests on streams, we suggest that most of these effects also apply to other waterbodies such as lakes and wetlands. A key question for determining whether forests are riparian is to ask whether the removal of the forest could measurably affect the adjacent aquatic environment. This defines "riparian forest" in functional terms by asking whether or not the presence of vegetation influences the condition of the aquatic environment. Thus we define riparian vegetation as: "Any vegetation adjacent to waterbodies, that if removed, could result in a measurable change in the physical, chemical or biological properties of the waterbody."

We also define two general, functional components to riparian corridors, an interaction buffer and an input buffer. (Figure 1). The interaction buffer is the stream and stream-adjacent buffer that regularly affects and is affected by fluvial processes. This represents the area where aquatic habitat currently exists or could be formed by natural processes. Therefore it is approximated by the area where a channel and its floodplain currently exist or are likely to be found in the foreseeable future. This area includes both aquatic and riparian habitat, but it is unique in the landscape in that any particular location may become aquatic habitat, and that many locations have a characteristic frequency with which they switch back and forth between aquatic and riparian habitat. For example, the location of a young riparian forest on a point bar on a meander bend along a river is the same location where the river once was, perhaps a few years earlier. There is a certain probability that at some point in time the river will meander back to that point. Thus over time, that particular location exists as both aquatic and riparian habitat.

In contrast, input buffers include areas that can or do deliver or regulate substantial material or energy inputs, primarily organic matter (e.g. LWD), sediment, and thermal energy, to streams, but are not directly affected by fluvial processes (e.g. erosion and deposition). Input buffers are adjacent to the interaction buffer and always exist only as riparian habitat. Their purpose is to ensure that current and future aquatic habitat is adequately protected and supplied with appropriate levels of material and energy inputs. Input buffers constitute the majority of the riparian network, while interaction buffers are relatively limited in space, but are disproportionately important because it is in this area that aquatic habitat is created, maintained and ultimately destroyed.

Within these buffers are functional zones. These zones provide important ecological goods and services to streams that are generally described as riparian functions. The width of these zones are defined by the area that cumulatively provides full functionality (we define full functionality as the width that will provides 100% of all known functions of riparian zones). Zones included within the interaction buffer are the floodplain, channel migration zone (CMZ), and beaver habitat zone (BHZ). The floodplain and CMZ overlap considerably, and for the sake of simplicity we refer to the area covered by the floodplain or the CMZ as the CMZ.

Functional zones in the input buffer include LWD recruitment, small woody debris recruitment, litterfall production, shade and windthrow zones. On streams where there is no CMZ or BHZ, there is no interaction buffer, only the active channel and the input



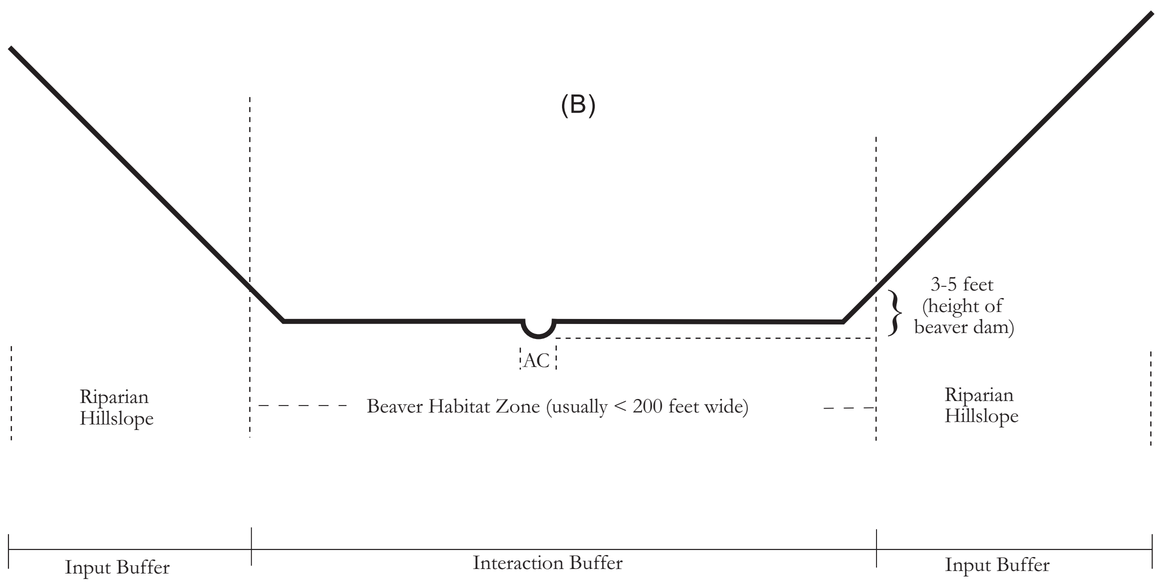


Figure 1. Visual representation of interaction and input buffers for (A) a meandering stream with a floodplain and (B) a small stream that could be dammed by beaver.The interaction buffer is the stream and stream-adjacent area that interacts with and is physically affected by fluvial processes. In contrast, input buffers are areas that regulate the input of energy and materials to the stream network, but are never directly affected by fluvial processes. Drawings are not to scale.

buffer. This situation describes the vast majority of riparian forests in the Pacific Northwest, since CMZs and BHZs are generally limited to low-gradient streams. All of these zones and how their functions vary with riparian width are discussed in separate sections, below.

FLOODPLAIN & CHANNEL MIGRATION ZONE

The floodplain and channel migration zone (CMZ) consists of the area that a stream and floodplain (including side channels) could potentially occupy under existing climatic conditions (Figure 1A). It is the area where there is a reasonable probability that aquatic or wetland habitat will exist at some time in the future. Such an area is often assumed to be approximately coincident with the 100 year floodplain, although it also includes lower terraces and hillslopes adjacent to the floodplain where the stream is likely to meander. The actual area where there is a reasonable probability that aquatic or wetland habitat will exist at some time in the future has not been well defined, and may include areas outside the 100 year floodplain. The CMZ is the physical location where aquatic (i.e. salmonid) habitat will be created at some point in the future and therefore should have the same level of protection as existing aquatic habitat.

The CMZ is a feature most extensively developed on larger, low-gradient streams where fluvial processes build floodplains. On higher gradient streams CMZs are usually absent, though they can be found anywhere there is sufficient LWD to create an alluvial bedform wider than bankfull width. Thus the CMZ is non-existent in small watersheds and greater than a mile wide in large drainages such as the Skagit River. For most watersheds regulated by Washington's forest practice regulations, the CMZ is generally limited to a very small fraction of the entire stream length.

In addition to providing a physical template for the creation of stream and wetland habitat, the portion of the CMZ currently covered by vegetation functions as a riparian buffer. Thus, important functions of living trees in the CMZ include providing a source of LWD, shade and cover, stabilizing off-channel habitat, contributing to bank stability with their root networks, fine and coarse sediment storage (from both overbank flow and upslope erosion), and providing organic material to aquatic habitat (reviewed in Naiman 1992). Primary functions of LWD in the CMZ include providing a future source of instream LWD as channels migrate, including the formation of log jams that regulate channel meandering and avulsions (Abbe and Montgomery 1996), and creating raised microsites for conifer establishment on flood prone areas (Harmon et al. 1986, Harmon and Franklin 1989, Beaudry et al. 1990, Naiman et al. 1992).

BEAVER HABITAT ZONE

The riparian area that is, or could be, flooded by the dam-building activities of beaver is the beaver habitat zone (BHZ). The BHZ is potential or existing aquatic habitat and is part of the interaction buffer (Figure 1B). The BHZ is of great importance to salmonids because the impoundments built by beaver create outstanding rearing and overwintering habitat, particularly for coho salmon, *Oncorhynchus kisutch* (Bustard and Narver 1975a, Bustard and Narver 1975b, Peterson 1982a, Peterson 1982b, Bryant 1984, Murphy et al. 1989, Leidholt Bruner et al. 1992). For coho salmon, the abundance of beaver ponds is an important determinant of the overall smolt productivity of a watershed (Pollock and Pess 1998). Riparian forests flooded by beaver are particularly beneficial because they combine two essential elements of good coho habitat; a productive and complex structural environment that provides cover (primarily from LWD) and low-velocity water.

Unfortunately, beaver populations are much lower than they have been historically and are kept low by continued trapping pressure (Naiman et al. 1988, WDFW 1997). Thus the amount of beaver-created habitat remains low, and the amount of aquatic habitat available to salmonids has been substantially reduced relative to natural conditions.

Beaver dams are generally found on small lakes and small streams and side channels with gradients less than 4%, although beaver often dam streams with gradients between 4-8% and sometimes as high as 16% (Retzer et al. 1956, Pollock and Pess 1998). The BHZ is the area that a typical, functioning beaver dam would likely to impound. In the Pacific Northwest, most beaver impoundments are less than 200 feet wide (Pollock and Pess 1998). Beaver prefer to dam small streams, with bankfull discharges less than 1 m^3s^{-1} and often dam streams with a bankfull discharge < 0.1 m^3s^{-1} (Pollock and Pess 1998). This suggests that in order to adequately protect salmonid habitat, small, low gradient streams need a buffer wide enough to accommodate the potential for a beaver pond. In the Pacific Northwest, typical beaver dams are 3-5 feet high, measured from the bed of the channel that they span (personal observation). Thus the BHZ can be delineated by a contour line along the valley side slope that is 3-5 feet above the channel bed. While most BHZs would be less than 200 feet wide, the exact dimensions would vary according to local topography.

Like other habitat in the interaction buffer, BHZ slowly pass through series of vegetational and aquatic stages (Johnston and Naiman 1990, Johnston et al. 1993), only some of which are beneficial to salmonids. However, beaver-modified habitat in all its successional stages is an important contributor to the biodiversity of a watershed (Pollock et al. 1998) and provides many other ecological functions (see reviews in Naiman et al. 1988, Pollock et al. 1994, Butler 1995). For both fish habitat and the maintenance of biodiversity, a desirable state for a beaver pond appears to be a flooded old-growth riparian forest (Bryant 1984, Pollock et al. 1998). There is enough topographical variation in an old-growth forest (as a result of the larger diameter woody debris, large stumps and pits adjacent to uprooted, windthrown trees) that a typical beaver dam does not actually flood the entire forest. Instead the flooding creates a complex forested swamp with channels, patches of open water and emergent vegetation interspersed among small forested islands and peninsulas containing one to several large conifers and a diverse understory of both forest and wetland plants. The complex spatial and temporal environmental gradients in such a small area explain why biodiversity is high, and also help to explain why such habitat provides excellent salmonid rearing opportunities. The habitat is extremely diverse in terms of topography, the availability of LWD, cover, types of detrital inputs, and degree of shading, thereby providing abundant food and protection against predators.

LARGE WOODY DEBRIS PRODUCTION ZONE

The importance of LWD (dead wood > 10 cm (4 in) dbh) to stream ecosystems, and the general absence of LWD in managed forest landscapes is well known. The dearth of LWD in most stream systems in Washington is generally thought to be one of the major reasons why so much freshwater salmonid habitat is severely degraded (Sedell and Luchessa 1981, Sedell and Froggart 1984, Bisson et al. 1987, Sedell et al. 1989, Sedell et al. 1993, Stouder et al. 1997, Naiman and Bilby 1998). Riparian forests are the source of almost all instream LWD. Thus the quality of instream habitat is largely determined by how the riparian forests have been managed. LWD influences the storage and routing of water, sediment, nutrients and organic material throughout channel networks and therefore plays an important role in determining both the quantity and quality of stream (and riparian) habitat. Specific functions of LWD important to salmonids on fish bearing streams include the formation of pools by altering channel hydraulics (e.g. concentrating flow), storing and sorting spawning gravels, providing cover, dissipating stream energy, capturing and concentrating nutrients and small organic material, and reducing bedload movement (Swanson and Lienkaemper 1978, Keller and Swanson 1979, Bilby and Likens 1980, Bilby 1981, Megahan 1982, Bilby 1984, Triska 1984, Triska et al. 1984, Lisle 1986, Bisson et al. 1987, Lienkaemper and Swanson 1987, Bilby and Ward 1989, Beaudry et al. 1990, Bilby and Ward 1991, Bilby and Bisson 1992, O'Conner and Harr. 1994, Montgomery et al. 1995, Haas 1996, Montgomery et al. 1996, Montgomery and Buffington 1997). On non-fish bearing streams, instream LWD benefits salmonids primarily by regulating the rate of sediment delivery to downstream reaches (Megahan 1982, Perkins 1989, Haas 1996, Montgomery et al. 1996) and by capturing and processing organic material, thereby increasing aquatic ecosystem productivity (Erman 1984, Triska 1984, Triska et al. 1984). Additionally, when debris flows occur, LWD is transported to fish bearing streams, thereby mitigating some of the effects of debris flows and helping to maintain salmonid habitat (Benda and Dunne 1987, Perkins 1989, Benda 1990, Benda and Cundy 1990, Benda and Sias 1998).

Theoretical considerations and empirical evidence suggest that under natural conditions most of the large woody debris entering reaches from the adjacent riparian zone originates from within one site potential tree height ($SPTH_{300}$) of the stream reach (McDade et al. 1990, Van Sickle and Gregory 1990). The relationship between buffer width and potential LWD inputs from the adjacent riparian zone is non-linear. For example, 95% of this LWD source comes from within a horizontal distance from the stream equal to 85% of the site potential (Figure 2). For low-gradient stream reaches, the relative importance of stream-adjacent sources of LWD versus upstream sources is less clear. Benda and Sias (1998) observed that debris flows are an important source of LWD to low-gradient streams. They recognized that instream LWD is created delivered via five processes: chronic low-level riparian stand mortality, episodic stand mortality caused by fires, bank erosion, channelized landslides (e.g. debris flows), and stream-adjacent hillslope landslides. The first three processes deliver LWD to a reach from stream-adjacent riparian stands, and thus the amount of LWD delivered from these sources can approximated from McDade's model. The latter two sources are from upstream or hillslope areas, and the amount of LWD delivered by these processes to a particular reach is less well known, but likely varies from watershed to watershed, depending primarily on geomorphology and climate. We are aware of no studies that quantify the relative amount of LWD entering streams from non-channelized landslides, while the only empirical study we know of that quantified the relative contribution of upstream LWD sources compared to stream-adjacent LWD sources was that of McGarry (1994).

McGarry (1994) determined that 48% of the LWD in the mainstem of Cummins Creek, a relatively pristine coastal Oregon stream, came from upstream sources, primarily debris flows (see also Burnett and Reeves 1997). The valley width of the mainstem of Cummins Creek is relatively narrow, so that debris flows entering the valley would have a high probability of hitting the mainstem, suggesting that 48% may represent a relatively high percentage of LWD entering low-gradient mainstem streams from high-gradient sources. In a wider (e.g. glaciated)

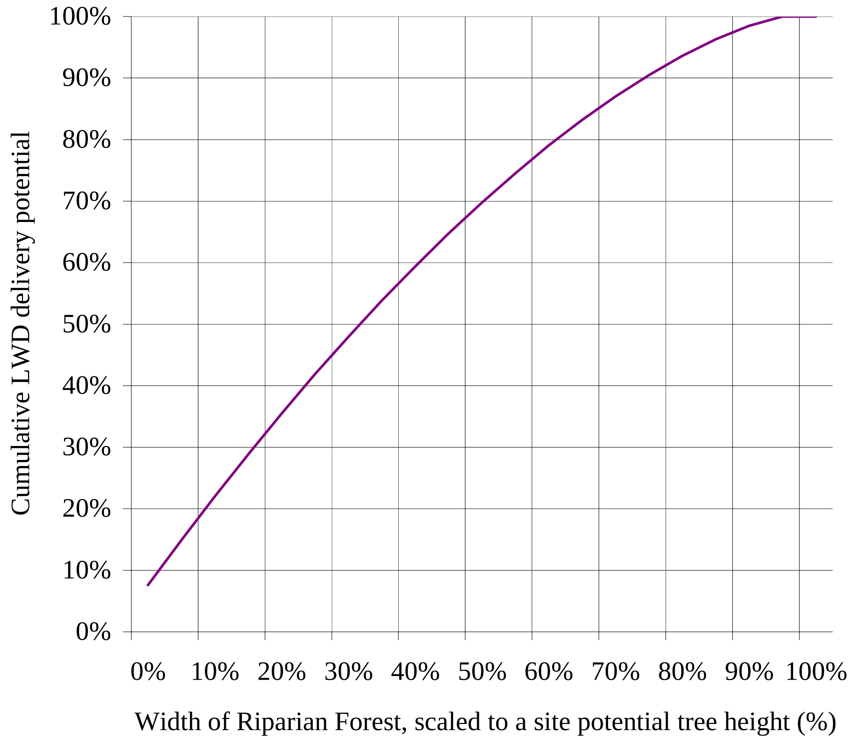


Figure 2. Cumulative instream LWD delivery potential of riparian forests as a function of distance from stream. Based on McDade et al. (1990).

valley, a number of debris flows would likely run-out on the valley floor without hitting the mainstem, so the relative contribution of debris flow-derived LWD to mainstem streams would probably be less. However these debris flows would follow stream channels and deposit LWD in and adjacent to smaller low-gradient channels crossing the valley floor, thus still contributing LWD to the fish bearing portion of the stream network and the CMZ.

Other researchers have found large concentrations of LWD at tributary junctions that were thought to have entered the low gradient channel from debris flows (see references cited in Burnett and Reeves 1997). Support also comes from McDade et al. (1990), who while estimating the point of origin of LWD in Pacific Northwest streams, could not determine the source of 48% of the pieces of LWD that they encountered. They comment that these pieces generally appeared more mobile than the pieces they identified as having come from stream-adjacent sources, indicating they may have been transported from upstream sources.

Together, these studies suggest that up to half of LWD in lower gradient (e.g. fish bearing) streams may come from upstream sources. This is an excellent example of longitudinal connectivity in riparian networks and lends support to the argument that salmonid habitat can only be protected in the context of the entire watershed (NRC 1992, Sedell et al. 1993, Stouder et al. 1997). In this example, if upstream LWD sources (i.e. riparian forests and unstable slopes) are not protected, the source of up to half of the LWD that would normally be delivered to low-gradient streams will be eliminated.

In managed forests, instream LWD recruitment potential is also affected by the presence of roads and yarding corridors in the riparian zone. The amount of riparian forests lost to yarding corridors may be at least 20% (e.g. proposed, *improved* Washington State Forest Practice regulations request that 20% of riparian corridors along all streams be left available for yarding corridors (Forestry Module 1998)). Additionally, many roads parallel the larger, low-gradient streams, further eliminating riparian forests. Just how much riparian forest has been converted into logging roads is not known. We have done some preliminary analyses (unpublished) using Geographic Information Systems data layers from DNR that suggests in watersheds dominated by commercial timber lands, 3% of riparian forests are currently occupied by road corridors. The exact amount of riparian forests lost to yarding and roads likely varies, but apparently constitutes a significant portion of the riparian zone. Based on the limited data available, we estimate that currently, 23% of riparian zones are lost to yarding corridors and roads.

Therefore, an estimate of the amount of potential LWD available to fish bearing streams in managed forests depends primarily on four factors: 1) the width of unlogged riparian forests along fish bearing streams, 2) the width of unlogged riparian forests on non-fish bearing (i.e. high gradient) streams, 3) the percentage of riparian corridors lost to yarding corridors and 4) the percentage of riparian corridors lost to roads. Other important factors include the riparian species and stand age (which determines tree height), the amount of LWD delivered by non-channelized landslides, and degree of lateral channel migration (see Benda and Sias 1998).

The long-term LWD recruitment potential for fish bearing streams can be estimated under various management scenarios (Figure 3), provided some simplifying assumptions are made. The first is that LWD delivered from non-channelized landslides is not a significant source of instream LWD and the second is that there is minimal channel migration (as is the case for most small streams). Additionally, these calculations estimate the LWD potential over the long-term, that is, when the trees have reached the height of old-growth forests. This assumption ensures a best-case scenario estimate of the amount of LWD that will eventually be available to streams under various management scenarios, assuming that all riparian zones eventually develop into old-growth buffers. In reality, in the interim between now and the time that buffers have reached old-growth like conditions, the amount of LWD available to streams will be somewhat less than our calculations suggest. Figure 3 provides estimates of the cumulative loss of LWD to fish bearing streams as a result of current and proposed forest practice regulations for typical westside forests. Additional examples, using differing assumptions, are provided in Table 1.

Under existing Forest Practice Rules (WFPB 1995a), the width of the riparian buffer varies with elevation, with more shade being required at lower elevations. For a westside watershed where fish bearing streams are on average 1000 feet above sea level, the average buffer width is 50 feet and therefore the LWD potential for such streams is about 20% (Table 1 and Figure 3). Proposed new rules may only slightly improve those numbers. For example, under riparian protection rules slightly better than those recently proposed in the Forestry Module (i.e. 50 foot, no cut buffers on 50% of the non-fish bearing streams, 100 foot no cut buffers on all fish bearing streams, 20% of all streams used as yarding corridors), the LWD recruitment potential for fish bearing streams improves to only 41%. These scenarios make it clear that existing and certain proposed (i.e. Forestry Module) regulations will provide nowhere close to fully functional riparian forests, even in the long-term. They also suggest that management activities within the riparian corridor contribute substantially to the loss of instream LWD recruitment potential, and that any plans to protect and restore salmonid habitat need to greatly reduce the extent of such activities.

Table 1. Long-term LWD recruitment potential for site class II and III forests under various management scenarios, relative to natural conditions. Standard FPRs (Forest Practice Rules) and the Forestry Module* allow yarding corridors to remove 20% of the long-term LWD sources, the DNR HCP allows a 10% loss, while SPTH on all streams and the WDFW WSP (Wild Salmonid Policy) assume no yarding corridors. For all management plans, roads are assumed to remove an additional 3% of the RMZ. Because of the uncertainty as to how much LWD in fish bearing streams comes from upstream sources, two sets of scenarios are included for each site class. The first assumes (based on McGarry 1994) that 50% of LWD comes from upstream sources (e.g. debris flows), while the second assumes that only 25% of LWD comes from upstream.The estimate for Current Standard Forest Practice Rules is for a watershed where the average elevation of fish bearing streams is 1000 feet and that therefore on average, a 50 foot shade buffer is required (WFPB 1995). DNR HCP and WDFW WSP scenarios assume that Type 4 streams constitute 30% of the non-fish bearing stream network, therefore the RMZ width given for non-fish bearing streams is a weighted average of Type 4 and all other non-fish bearing streams.

Proposal	Fish bearing RMZ width (feet)	Average non-fish bearing RMZ width (feet)	LWD recruitment potential relative to natural conditions (%)	Riparian assumed lost to yarding corridors	Riparian LWD assumed lost to roads	LWD of coming from non-fish bearing streams	LWD from RMZ adjacent to fb waters relative to total fb RMZ LWD potential	Potential LWD from fb RMZs relative to total LWD potential for fb streams	Potential LWD from nfb RMZs relative to total nfb LWD potential	Potential LWD from nfb RMZs relative to total LWD potential for fb streams
Site Class II, 25% of LWD comes fom non-fish bearing streams										
SPTH on all streams	215	215	97%	0%	3%	25%	100%	75%	100%	25%
Forestry Module*	100	25	39%	20%	3%	25%	62%	47%	18%	5%
Current Standard FPRs	50	0	20%	20%	3%	25%	34%	26%	0%	0%
DNR HCP	170	30	65%	10%	3%	25%	92%	69%	22%	6%
WDFW WSP	170	65	77%	0%	3%	25%	92%	69%	43%	11%
Site Class II, 50% of LWD comes fom non-fish bearing streams										
SPTH on all streams	215	215	97%	0%	3%	50%	100%	50%	100%	50%
Forestry Module*	100	25	31%	20%	3%	50%	62%	31%	18%	9%
Current Standard FPRs	50	0	13%	20%	3%	50%	34%	17%	0%	0%
DNR HCP	170	30	50%	10%	3%	50%	92%	46%	22%	11%
WDFW WSP	170	65	65%	0%	3%	50%	92%	46%	43%	22%
Site Class III, 25% of LWD comes from non-fish bearing streams										
SPTH on all streams	180	180	97%	0%	3%	25%	100%	75%	100%	25%
Forestry Module*	100	25	45%	20%	3%	25%	72%	54%	20%	5%
Current Standard FPRs	50	0	23%	20%	3%	25%	40%	30%	0%	0%
DNR HCP	150	30	66%	10%	3%	25%	93%	70%	25%	6%
WDFW WSP	150	65	80%	0%	3%	25%	93%	70%	50%	13%
Site Class III, 50% of LWD comes fom non-fish bearing streams										
SPTH on all streams	180	180	97%	0%	3%	50%	100%	50%	100%	50%
Forestry Module*	100	25	35%	20%	3%	50%	72%	36%	20%	10%
Current Standard FPRs	50	0	15%	20%	3%	50%	40%	20%	0%	0%
DNR HCP	150	30	51%	10%	3%	50%	93%	47%	25%	13%
WDFW WSP	150	65	69%	0%	3%	50%	93%	47%	50%	25%

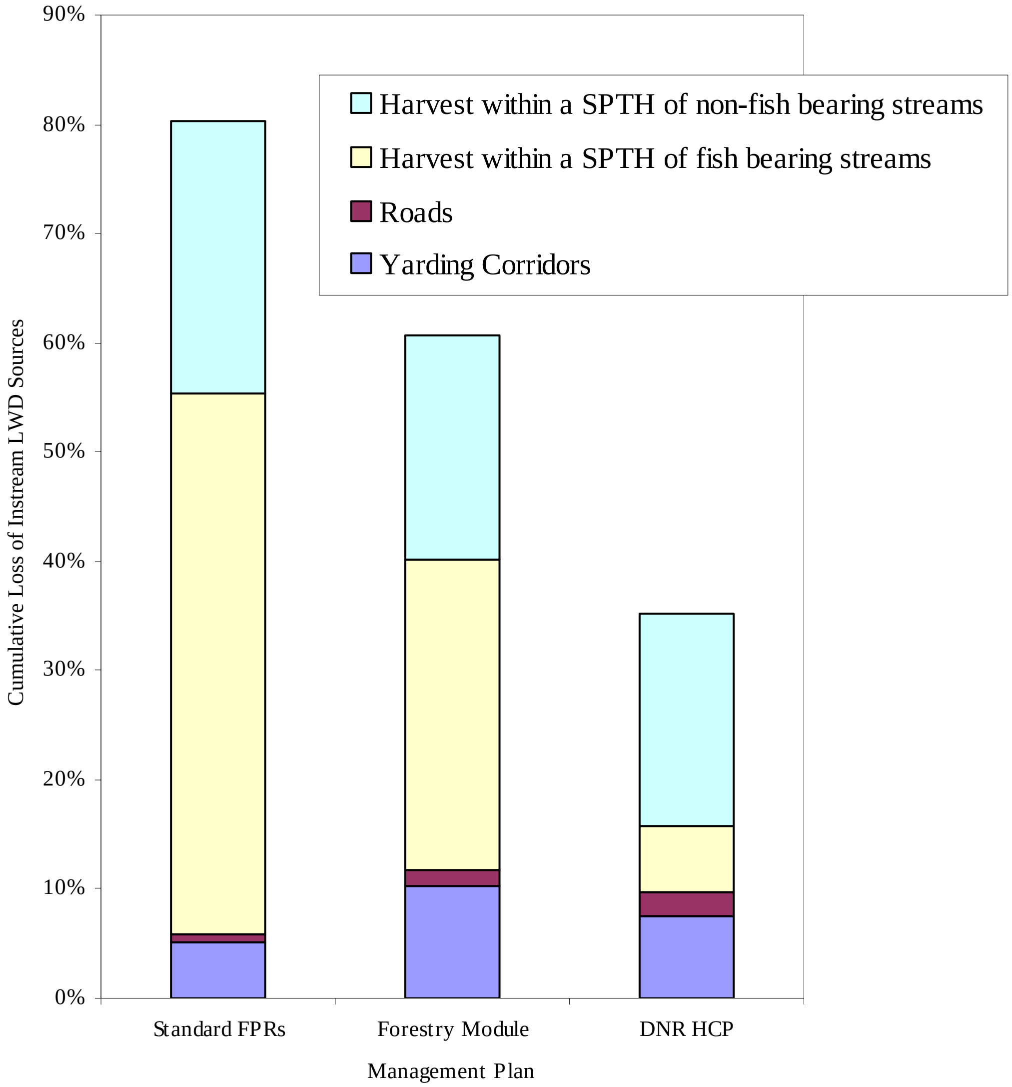


Figure 3. Cumulative loss of LWD sources for fish bearing streams under various management scenarios, relative to natural conditions. Standard FPRs (Standard Forest Practice Rules) scenario is for a watershed with an average elevation of 1000 feet for fish bearing streams and therefore an average riparian shade buffer width of 50 feet on fish bearing streams and no buffers on non-fish bearing streams. Yarding corridors are assumed to remove 20% of the trees within the buffer and roads are assumed to remove another 3%. Forestry Module* scenario assumes 100 foot buffers on fish bearing streams and 50 foot buffers on 50% of non-fish bearing streams. Yarding corridors are assumed to remove 20% of the trees within the buffer and roads are assumed to remove another 3%. DNR HCP (Department of Natural Resources Habitat Conservation Plan) scenario assumes 170 foot buffers on fish bearing streams, 100 foot buffers on Type 4 streams and no buffers on all other streams. Yarding corridors are assumed to remove 10% of the trees within the buffer and roads are assumed to remove another 3%. All calculations are for riparian buffers with a SPTH of 215 feet (a 100 year site potential of 170 feet), and assume that 25% of LWD in fish bearing streams originates from upstream (non-fish bearing) reaches. For comparative purposes, it is assumed that no harvest occurs within the riparian buffers, therefore the figure illustrates minimum LWD loss estimates.

SMALL WOODY DEBRIS PRODUCTION ZONE

Branches and other woody material not classified as LWD are considered small woody debris (SWD). SWD is an important component of the wood budget of forested streams, accounting for about 30% of the total wood volume (McDade 1987, cited in Van Sickle and Gregory 1990). Small woody debris is important to fluvial ecosystems because it helps create and maintain pool habitat in small and medium-sized streams. Increases in the frequency of debris-associated pools in old growth streams (W_{BF} = 10-75 feet) are due not only to increased levels of LWD relative to managed systems, but are also the result of increased levels of SWD (Bilby and Ward 1991). Streams within old-growth forests not only have more pools, but also a greater diversity of pools relative to managed systems, in part because they contain larger volumes of SWD. Plunge and dammed pools are often associated with accumulations of SWD. Such debris fills in the gaps of the framework provided by LWD, resulting in more efficient blockage of stream flow. Natural SWD accumulations are generally rare in managed forests and so are plunge pools (Bilby and Ward 1991). The amount of SWD associated with LWD may also influence the formation of depositional sites. LWD with large SWD accumulations retains sediment significantly more frequently than LWD which has low accumulations of SWD. Available information also suggests that the amount of SWD decreases at a rapid rate following removal of riparian vegetation. SWD levels decrease by about 90% a few years after harvest and remain at low levels for at least half a century (Bilby and Ward 1991).

In summary, this information suggests that both the abundance and diversity of pools in small forested streams is dependent not just on LWD, but on SWD as well, and that management activities in riparian zones which remove potential SWD sources to stream networks contribute to the degradation of instream (e.g. salmonid) habitat, particularly for small streams.

An estimate of how the potential for SWD delivery from riparian areas varies as a function of distance from the stream by calculating the probability that a branch at a particular height on a tree a particular distance from the stream will fall into the stream during treefall (Pollock et al., In preparation):

$$Pcs = (\mathrm{asin}((z+cw)/bh)/180 - \mathrm{asin}(z/bh))/180$$

Where:
Pcs = probability of a branch in a falling crown being delivered to a stream, bh = branch height, z = horizontal distance from tree base to stream channel, and cw = channel width.

The model assumes that riparian treefall direction is random and that the density, height and rate of treefall throughout the riparian zone are randomly distributed. In addition, the model assumes that all SWD comes from the crowns of trees, that all trees have a crown 60 feet deep (Tim Beechie, personal communication), and that branches in tree crowns are randomly distributed. This model was used to compute a cumulative probability distribution of SWD sources from the riparian zone (Figure 4).

The model produces a sigmoidal curve of SWD recruitment, and suggests that for small streams ($\leq$ 10 ft W_{BF}), more than half the potential instream SWD is in from the outer half of the riparian zone (where the riparian zone is defined as one site potential tree height distance from the bankfull channel). As streams get larger, the curve becomes less sigmoidal and more convex, and for streams with a bankfull channel width greater than a site potential tree height, the SWD recruitment curve becomes similar to the convex LWD recruitment curve of McDade (compare Figure 4 with Figure 2). The difference between the shapes of the SWD and LWD recruitment curves for small streams stems from the fact that when large trees close to the stream edge fall, their crowns are more likely to

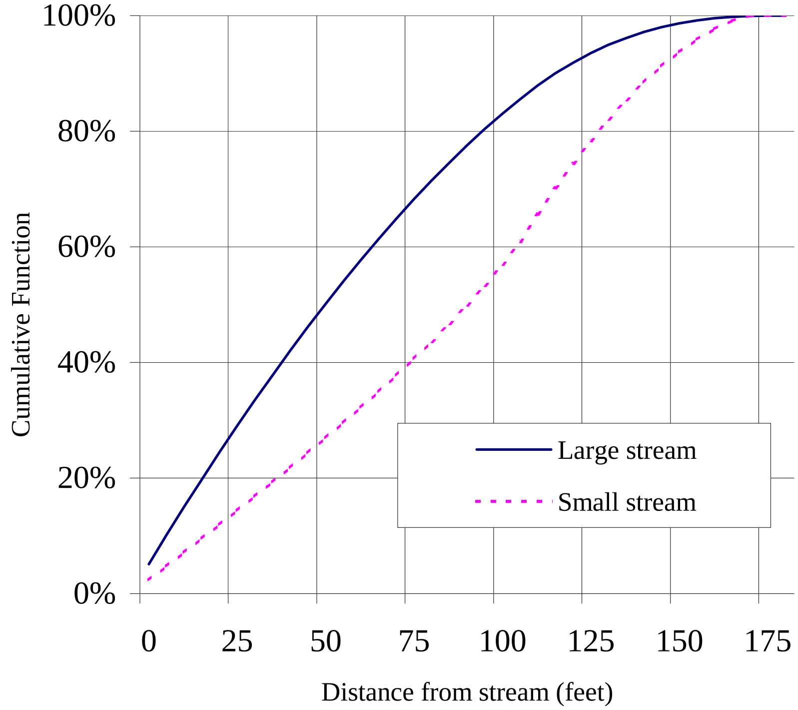


Figure 4. Small woody debris recruitment curves for small and large streams in site class III forest land, where SPTH = 170 feet.

land in the opposite riparian zone rather than in the channel.

Like all models, the accuracy of the results rests in large part on the validity of the assumptions. McDade et al. (1990) speculated that closer to the stream, tree densities and mortality rates may be higher, but provided no supporting data. We hypothesize (with no supporting data) that:
1) Trees adjacent to streams on the edge of the riparian forest probably have larger crowns than trees in the middle of the forest because of the increased light availability, 2) many branches in old-growth forests are LWD not SWD, and 3) direct delivery (i.e. not by tree fall) of SWD from stream-adjacent trees are important. However, until these ideas are tested, our model presents the best estimate of how SWD delivery from riparian forests varies as a function of distance from the stream.

The model does not quantify the relative importance of upstream sources of SWD relative to stream-adjacent sources. However, existing studies of LWD sources suggest that close to half the instream LWD comes from upstream (i.e. tributary) reaches (McGarry 1994) or could not be identified as coming from stream-adjacent areas (McDade et al 1990). Because of the higher mobility of SWD relative to LWD, we suggest that for a given reach, the SWD contribution of upstream reaches will also likely be higher than for LWD. However, this needs verification.

In summary, available data and the model results suggest that for small streams, SWD plays an important role in forming pools, trapping sediment, and structuring the biological communities, and that most SWD comes from the outer half of the riparian zone. In the case of many western Washington forests, this suggests that under natural conditions, the majority of SWD delivered to small streams comes from the portion of the riparian forest further than 100 feet from the stream edge. As streams become wider, the SWD recruitment curve slowly changes shape until at a channel width equal to a site potential tree height, the curve is approximately the shape of the LWD recruitment curve of McDade (1990).

LITTERFALL PRODUCTION ZONE

Fine organic material produced by trees and other vegetation not classified as LWD or SWD is called litterfall. This primarily includes leaves, needles and small twigs. This detritus decomposes relatively quickly and is important for stream systems because it provides food and energy to the aquatic food web. This directly boosts production of benthic invertebrate detritivores, and indirectly increases production of predators such as salmonids, which feed on these organisms (see review in Gregory et al. 1987). The temporal patterns of deciduous and coniferous litterfall to streams are different. On an annual basis, deciduous litterfall enters streams in small amounts over the course of the growing season, and culminates in a pulse of inputs to the stream in the autumn during leaf fall. Because deciduous litterfall primarily comes from leaves falling off of standing trees, such material probably comes from trees fairly close to the stream. In contrast to deciduous trees, coniferous litterfall enters the stream in small amounts over the entire year and large pulses occur during heavy windstorms, usually in the winter, when wind topples trees and small branches and needles are stripped from standing trees. We are not aware of studies quantifying the relative importance of coniferous litterfall to streams from windthrown trees versus direct inputs from standing conifers, or any studies that have measured the source distance of litterfall in streams. Therefore, we defer to the qualitative litterfall recruitment curve described in FEMAT (1993, Figure V-12), which suggests that most litterfall comes from within about half of a $SPTH_{300}$ (see also Erman et al. 1977). Until additional data are gathered, we suggest this is the best estimate of the litterfall recruitment curve from riparian forests.

SHADE ZONE

Maximum stream temperatures in forested watersheds are strongly influenced by the presence of shade from riparian forests (e.g. Brown and Krygier 1970, Brazier and Brown 1973, Beschta et al. 1987). Also important is the temperature of groundwater entering stream systems (Constantz et al. 1994, Constantz (in preparation)), which can be influenced by riparian (and upland) forest conditions as well (Brosofske et al. 1997, Olson 1998). During the summer, direct solar radiation to streams is a primary factor influencing temperature increases. Therefore partial or complete removal of riparian canopies providing shade to streams directly influences (increases) summertime stream temperatures. The relationship between riparian buffer width and the amount of shade provided to streams relative to natural conditions has been quantified by several research teams. Data collected by Steinblums et al. (1984), describes the relationship between Angular Canopy Density (ACD) and buffer width for forested streams in western Oregon. Their data suggest that the amount of shade provided by riparian trees rises exponentially with buffer width, and reaches 100% of natural conditions at approximately 140 feet (Figure 5). Similarly, Brosofske et al. (1997) also described the relationship between solar radiation received by streams and buffer width for streams in western Washington. However, their data describe a less rapidly rising exponential curve, and suggest that 100% of natural shade levels is not reached until approximately 250 feet (Figure 5).

The curves are relatively similar for narrow (< 50 foot) buffers, and both curves suggest that 50 foot buffers only provide about half of the shade as compared to a natural stand. For further comparison, the ACD curve suggests that 100 and 130 foot buffers provide shade equal to 85% and 95% of

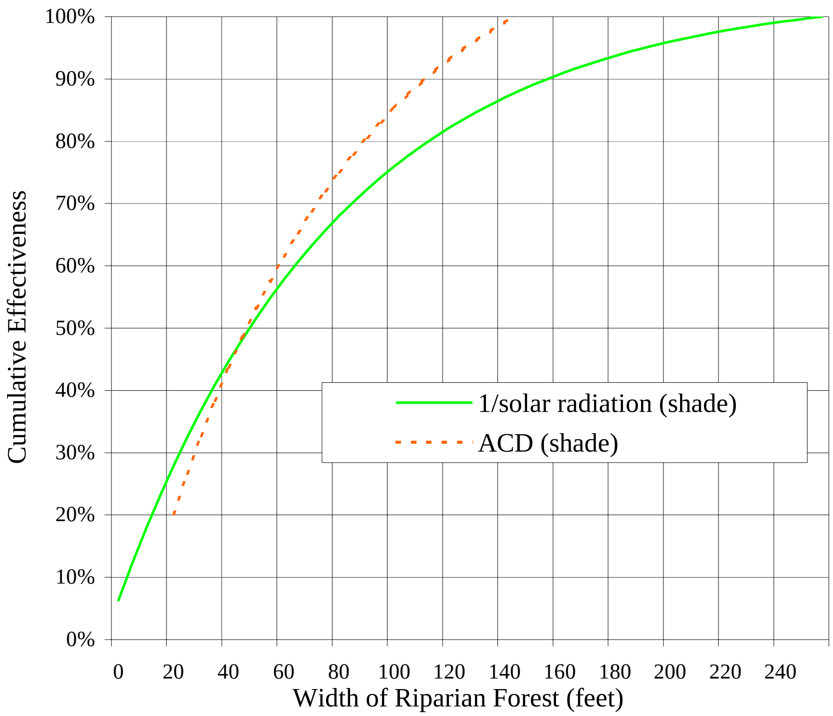


Figure 5. Comparison of estimated benefits of riparian forests to stream shading as a function of distance from stream from measuring solar radiation and angular canopy density (ACD). Based on Steinblums et al. 1984, Brosofske et al. 1997.

natural conditions, respectively, whereas the solar radiation curve suggests that buffer widths of 140 feet and 190 feet are needed to achieve similar shade levels. The differences between the curves arise in part because ACD is a visual estimate of the amount of shade provided, whereas solar radiation is directly measured using instrumentation (e.g. a LI-COR silicon pyranometer). Because measures of ACD require a visual estimation subject to user interpretation, we believe that the direct measure of solar radiation provides a more accurate estimate of the amount of solar radiation received by streams. Therefore we consider the data provided by Brosofske et al. (1997) to be the better estimate of the relationship between riparian buffer width and the amount of solar radiation reaching streams

While relationships between riparian buffer width and shade at the stream are reasonably clear, the relationship between the width of a riparian buffer along a particular stream reach and the stream temperature in that particular reach is less clear. Early research determined that narrow riparian buffers greatly lowered temperatures in the adjacent stream when compared to streams with no buffers at all (e.g. Brown and Krygier 1970). Since then, attempts have been made to develop quantitative relationships between shade and other parameters with stream temperature, but with little success. The most comprehensive of these attempts was the study by Sullivan et al. (1990) who examined data from 47 streams throughout Washington. In their study, they report that shade alone could explain only 15% of the variation in stream temperature and that no one factor alone was a good predictor of stream temperature. They also developed a model to predict maximum stream temperatures that included shade, elevation, mean air temperature, discharge and W_{BF} as independent variables. This model could predict 69% of the variation in maximum stream temperatures. However, mean air temperature could not be removed from the model without a significant loss of model reliability. Nevertheless, the authors concluded that shade and elevation alone could be used to determine the width of riparian buffer needed to meet State water quality standards. To bolster this assertion they presented some quantitative methods for making a rough estimate of the amount of shade needed on a particular stream for a given elevation in order to meet these standards (Sullivan et al. 1990, Figures 7.8 & 7.9). However, since most (79%) of the streams they examined had maximum temperatures above the State water quality standard for type AA streams, it is not clear how they determined the conditions under which water quality standards would be met. Our interpretation of their data is that they have presented a set of shade and elevation conditions where water quality standards are not likely to be met. This does not mean that outside of those conditions, water quality standards will necessarily be met.

In contrast, Hatten and Conrad (1995), in a study of unmanaged (< 15% of watershed harvested) and managed (>15% harvest) low elevation (< 850 feet) watersheds on the Olympic Peninsula, concluded that elevation and shade were not the most important variables for estimating stream temperatures. Of particular interest was the fact that for streams with similar shade levels in both unmanaged and managed watersheds, the unmanaged streams had significantly cooler temperatures. The average shade level for managed and unmanaged streams was 65% v. 72%, respectively. (Since the shade measures were taken with a densiometer, Steinblum's (1984) curve (see Figure 5) provides a rough estimate of the average buffer width, which would be about 70 feet and 80 feet for managed and unmanaged streams, respectively). Also, a minority (36%) of the streams in the lightly (un)managed watersheds met State water quality standards, while almost all (93%) of the streams in the (heavily) managed watersheds did not meet water quality standards, even though the average amount of shading was similar for both sets of streams. Therefore, these data suggest that some parameter(s) other than direct shade strongly influences stream temperatures.

In their study, Hatten and Conrad also examined data quantifying the watershed area covered by late-seral forests. This variable was a reasonably good predictor of temperature, explaining 52% of the site to site variation in maximum stream temperatures. They used the area of late-seral forests as an index of the amount of logging that had occurred, and concluded that there was a cumulative effect from logging that raised stream temperatures, even when buffers were left along (fish bearing) streams. Other studies generally support the conclusion that increases in stream temperatures are correlated with the amount of logging activities that have occurred in a watershed (Beschta and Taylor 1988, Holtby 1988). These studies suggest that temperature in a given stream reach is affected not only be the condition of the adjacent riparian forest, but by riparian and hillslope conditions far upstream and upslope. Under existing regulations at the time of these studies, it is likely that in the watersheds where these studies were conducted, few buffers were left on non-fish bearing streams. Therefore, it is difficult to know to what extent these cumulative temperature effects could have been avoided by providing adequate buffers on non-fish bearing streams while still logging adjacent hillslopes. However, since sunlight is the primary energy source that heats streams, it seems reasonable to suggest that providing shade to all streams would be an appropriate first step towards eliminating these cumulative effects. Whether or not that alone would eliminate the cumulative temperature effects of logging is not known.

Groundwater temperatures also influence stream temperatures. In some situations, clearcut logging may affect groundwater temperatures, suggesting that the (cumulative) impacts of forest practices on groundwater temperatures also need to be examined. Where groundwater is close to the surface, removal of the forest canopy may increase groundwater temperatures. For example, Hewlett and Fortson (1982) suggested that narrow riparian buffers had little effect on the temperature of some streams in the southeastern United States because forests had been cleared above shallow aquifers. Brosofske et al. (1997) showed a strong relationship between upland soil temperatures and stream temperatures, concluding that clearcuts outside the riparian zone may increase stream temperatures by raising groundwater temperatures. Olson (1998) modeled the potential for upland forest removal to affect groundwater temperatures. The study indicated that the affect of forest removal on groundwater temperatures was dependent on the physical characteristics of the aquifer (e.g. porosity, conductivity etc.) and the proximity of the groundwater to the surface. If the assumptions of the model are reasonably accurate, the results suggest that forest removal can affect groundwater temperatures up to a depth of six feet.

In summary, the studies reviewed suggest that 70-80 foot riparian buffers applied to fish bearing streams is not sufficient to maintain stream temperatures that approximate natural conditions, or even meet State water quality standards. The studies suggest that cumulative heating of fish bearing streams is occurring because: 1) insufficient shade is provided to non-fish surface waters and 2) shallow groundwater supplies are warmed when upland forests are cleared. There may also be other ways in which forest practices increase stream temperatures that we have yet to discover. That is, even if adequate shade is provided to all streams and timber harvest is restricted over upland areas with shallow aquifers, it is still uncertain as to whether this will allow stream temperature regimes to return to natural conditions. For example Beschta and Taylor (1988) observed that widespread mass wasting caused be logging operations can widen numerous streams, and therefore might contribute to cumulative increases in stream temperatures.

However, given the data currently available, we believe that if riparian forests were wide enough such that the quantity of solar radiation reaching all perennial (i.e. not dry in the summer) surface waters and shallow groundwater areas in forested watersheds were reduced to natural levels, that the condition of the riparian forest itself would no longer be contributing to increases in stream temperatures.

WINDTHROW PROTECTION ZONE

If riparian buffers are going to remain functional over the long-term, they need to be relatively windfirm. Several studies have attempted to define the relationship between riparian windthrow in the Pacific Northwest and various physical features such as topography, valley morphology, aspect, slope, etc. (Steinblums 1978, Steinblums 1984, Harris 1989, Sherwood 1993, Mitchell 1995, Mobbs and Jones 1995, Sinton 1996, Rot and Naiman (Submitted)). No one factor was found to be of particular importance, although Steinblums (1984) was able to develop a multiple regression using 7 independent variables that explained 74% of the wood volume loss that occurred as a result of windthrow. However, we reanalyzed several of these data sets (Sherwood 1993, Mobbs and Jones 1995, Rot and Naiman (Submitted)), looking at the relationship between buffer width and the likelihood of windthrow and reached the simple conclusion that forests in narrow streamside buffers ($\leq$75 ft) have a much higher probability of suffering appreciable mortality from windthrow than forests in wider buffers.

Riparian windthrow data from the Olympic Peninsula indicate that 28% of buffers less than 75 feet wide experienced appreciable windthrow mortality (5-50% of all trees) within a few years of harvesting the adjacent upland forest (Mobbs and Jones 1995). This contrasts with buffers wider than 75 feet, which had appreciable mortality in just 12%

of the buffers sampled (Figure 6). Other studies of riparian buffer survival generally support these conclusions. For example, Rot and Naiman (Submitted) observed that 1-2 years after harvesting adjacent uplands, blowdown in riparian buffers 50-115 ft wide was between 0-40% of all trees, whereas mortality in buffers less than 50 ft wide ranged from 0-100%, with over half those sites experiencing greater than 40% mortality.

Over longer time periods (decades), the difference between the relative stability of narrow and wide buffers continues to increase. We reanalyzed the combined data sets of Steinblums (1978) and Sherwood (1993) who both examined the same 20 riparian buffer strips in the west Cascades of Oregon 13-15 years apart. Our analysis indicates that about three-fourths of riparian buffers less than 80 feet wide experience significant blowdown (> 20% volume or basal area), while only 14% of wider buffers lost an appreciable number of trees.

Steinblums et al. (1984) from 1975-1977, examined 40 buffers strips that were between 1-15 years old. Sherwood (1993) selected 20 of these buffer strips for reinventory in 1990, at which point they were between 16-30 years old. However, one site could not be relocated and another had been selectively logged. Of the remaining 18 buffers, 11 were initially between 30-80 feet wide, while 7 buffers were 110-190 feet. Steinblums original data showed that 5 out of 12 of the narrow buffers experienced appreciable volume losses due to windthrow (>33% of the original stand volume), whereas only 1 of the wide buffers had appreciable amounts of windthrow. Sherwood's (1993) data indicate that in the 15 years between studies 3 more narrow buffers experienced windthrow losses greater than 20% of the basal area, and 4 of the 5 narrow buffers that had windthrow at the time of Steinblums' study continued to have substantial, additional windthrow losses (Sherwood measured changes in basal area, so the results are not directly comparable to Steinblums (1978) volume estimates). In contrast, no wide buffers had windthrow losses greater than 20%.

Therefore, of the 18 buffers studied by both Steinblums (1978) and Sherwood (1993), 8

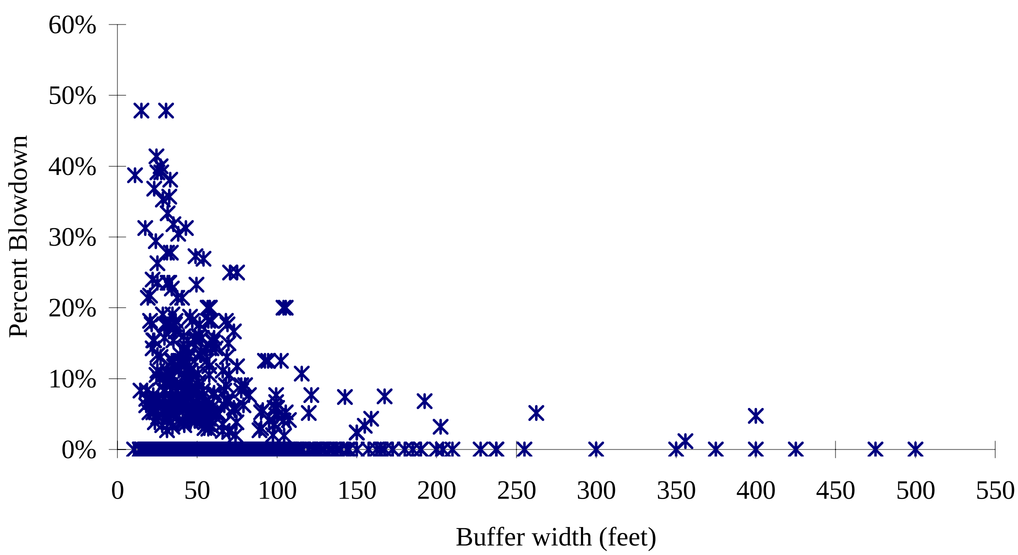


Figure 6. Riparian buffer width vs. percent of tree blowdown within the buffer (adapted from Mobbs and Jones 1992).

out of 11 (73%) of the buffers less than 80 feet wide suffered blowdown, whereas 1 out of 7 (14%) of the buffers wider than 110 feet suffered blowdown (Table 2). These studies, in combination, provide the only long-term data on riparian buffer survival of which we are aware. They suggest that wider buffers (in this case 110 feet or more) are generally windfirm, whereas buffers less than 80 feet wide have a high probability (73%) of experiencing moderate to severe windthrow over long periods (decades).

In summary, these studies suggest that beginning in the range of 75-110 feet, buffers generally become windfirm, and losses due to windthrow are generally low. In contrast, narrow buffers less than 75 feet wide have a much higher probability of suffering windthrow, particularly over the long-term. In the first few years after harvest, windthrow may damage slightly less than a third of such buffers, while over the long-term, 73% of these narrow buffers may be damaged. This suggests that 75 feet (and possibly upwards to 110 feet) constitutes the minimum buffer width which can be expected to incur minimal windthrow losses over the long-term.

Table 2. Relationship between riparian buffer width and windthrow in forests of the western Oregon Cascades. Data derived from Sherwood 1993 and Steinblums 1978 as cited in Appendix A of Sherwood 1993. For Steinblum's data, wood volume losses > 20% were considered to be windthrow. For Sherwood's data, basal area losses >20% were considered to be windthrow.

Year of data collection	RMZ width (ft)	Sample size (n)	Number of sites with windthrow by 1977	Number of additonal sites with windthrow by 1990	Total number of sites where windthrow occurred	Percentage of sites where windthrow occurred
1975-1977	40-80	11	5	3	8	73%
1990	110-190	7	1	0	1	14%
Total		18	6	3	9	50%

MICROCLIMATE PROTECTION ZONE

Riparian vegetation protects stream corridors against climatic changes caused by widespread land use activities such as forest removal (i.e. clearcutting) outside the riparian corridor. Important microclimatic parameters that riparian vegetation helps modulate include soil, and air temperature, humidity and wind speed. Microclimate conditions have not yet been directly linked to the condition of salmonid habitat. However, we suggest that stream (and riparian) microclimate needs protection in order to protect and restore stream environments to more natural conditions. Microclimate is known to be important for stream/riparian species other than fish (e.g. riparian plants and riparian-dependent wildlife), and may also influence water quality, particularly temperature (Sullivan et al. 1990, Petranka et al. 1993, Brosofske et al. 1997). To the extent that other species are important for maintaining functional aquatic systems, they are also important to salmonids. Salmonids are adapted to and evolved in aquatic systems containing certain species. Removal or a reduction in the abundance of these species through alterations to microclimate or otherwise,

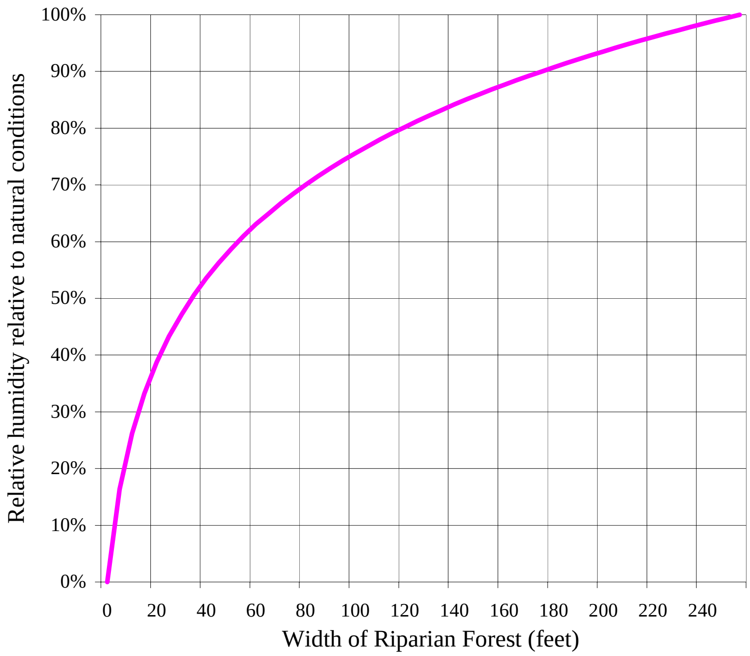


Figure 7. Cumulative benefits of riparian forests to regulating relative humidity near streams as a function of distance from stream. Based on Brosofske et al. (1997).

degrades aquatic habitat, and represents an unknown risk to salmonids.

The relationship between riparian width and microclimate protection varies with the parameter being examined, and quantitative relationships between buffer widths and many microclimate variables have not been developed. Chen et al. (1995) examined the effects of clearcuts on nearby forest interiors. They concluded that clearcuts could influence microclimate conditions far into the forest interior, and that the effects of clearcuts on the adjoining forest microclimate were greatest for southwest facing sites. For example, changes in relative humidity could be measured 30-240 m into the forest interior, and were greatest on south facing sites. Changes in soil temperature could be measured 60 m into the forest and were greatest on south facing sites, while soil moisture was affected 90 m into the forest on south-facing sites, with no measurable effect on all other orientations.

Brosofske et al. (1997) directly measured the effects of riparian buffer width on stream microclimate. They found no obvious relationships between buffer width for air temperatures or wind speed near the stream, but did find quantifiable relationships between buffer width and solar radiation (discussed under the shade section) and buffer width and relative humidity. The quantitative relationship between relative humidity and buffer width is described in Figure 7, and demonstrates that a buffer width of approximately 250 feet is needed to maintain a relative humidity at the stream that approximates natural conditions. A riparian buffer this wide also should ensure that most other microclimate parameters such as air temperature, soil temperature, and surface air temperature are not substantially altered around the stream environment, relative to reference conditions, under most situations (Brosofske et al. 1997, Chen et al. 1995).

A 250 foot buffer should maintain natural microclimate conditions at the stream edge only, but not in the riparian forest itself. Therefore riparian management designed to maintain a natural microclimate for riparian-dependent plants and animals within portions of the riparian forest will require wider buffers, or essentially, a buffer for the riparian environment.

Recommended riparian buffer widths

Given the imperiled status of many of the salmonid stocks in Washington, we recommend buffer widths that minimize the risk of further degradation to freshwater aquatic habitat resulting from management activities in riparian areas. These buffers are intended to minimize the future risk of extinction to salmonids caused by future timber harvest in riparian corridors to the furthest extent possible, given our current state of knowledge. We view any buffers designed to provide less than 100% riparian functionality as an unquantified risk to salmonid populations. Therefore, based on the data we have reviewed (summarized in Figure 8), this suggests interim buffer widths of at least 250 feet on all perennial streams and one site potential tree height on all seasonal streams are appropriate to ensure as close to 100% riparian functionality as is reasonably possible over the long term (Figure 8). The inner edge of this buffer would begin at the edge of the bankfull channel, the floodplain, channel migration zone or beaver habitat zone, whichever is wider. Such buffers ensure that the amount of organic material (LWD, SWD and litterfall) delivered to streams, the amount of shade provided to streams and the relative humidity at the stream will eventually return to approximately natural levels, and that the buffer is windfirm. A buffer width of 250 feet (as opposed to a $SPTH_{300}$) is needed on perennial streams primarily to maintain natural shade and humidity levels at the stream (Figure 8). In contrast, seasonal streams by definition, do not have water during the summer months when shade and humidity are a concern. Therefore, buffers on these streams can be reduced down to a $SPTH_{300}$.

These riparian buffers are not designed to ensure that sediment delivered to streams eventually returns to natural levels. Most sediment entering stream networks in forested Pacific Northwest watersheds is delivered by mass wasting. Mass wasting caused by logging-related activities on unstable slopes is a serious problem that degrades both instream and riparian habitat. Most riparian buffers can not fully mitigate the effects of debris flows. Therefore, no amount of riparian protection will allow salmonid habitat to recover unless there is concurrent protection for unstable slopes.

We consider the width of these riparian buffers widths interim because they will change as additional research and experimentation provides data that demonstrate under what conditions different sized buffers will provide fully functional riparian forests. Additional data will likely indicate that the width of fully-functional riparian forests will vary according to site-specific conditions. For example, more site-specific relationships between solar radiation received by streams and buffer width should be developed because the quantity of solar radiation received by streams is also affected by valley topography. Since topographic features have the potential to provide some degree of shading, collecting additional data will likely result in narrower buffers for certain valley types. Additionally, empirical evidence that narrower buffers do not lead to increases in stream temperatures could also result in changes in required buffer widths. Conversely, data regarding the effect of timber harvest over shallow groundwater aquifers may lead to wider buffers. The concern in this case is the potential negative effect that such harvest activities have on the thermal regimes of streams (Hatten and Conrad 1995). Data generated on this issue may not only result in wider riparian buffers, but also the protection of areas that are not necessarily adjacent to streams.

In the absence of a quantitative risk assessment (discussed below), no amount of data will provide a rationale for buffer widths that are less than one $SPTH_{300}$ wide. This is because one $SPTH_{300}$ is the width that will ensure that the amount of LWD, SWD and litterfall received by streams is approximately equal to natural conditions.

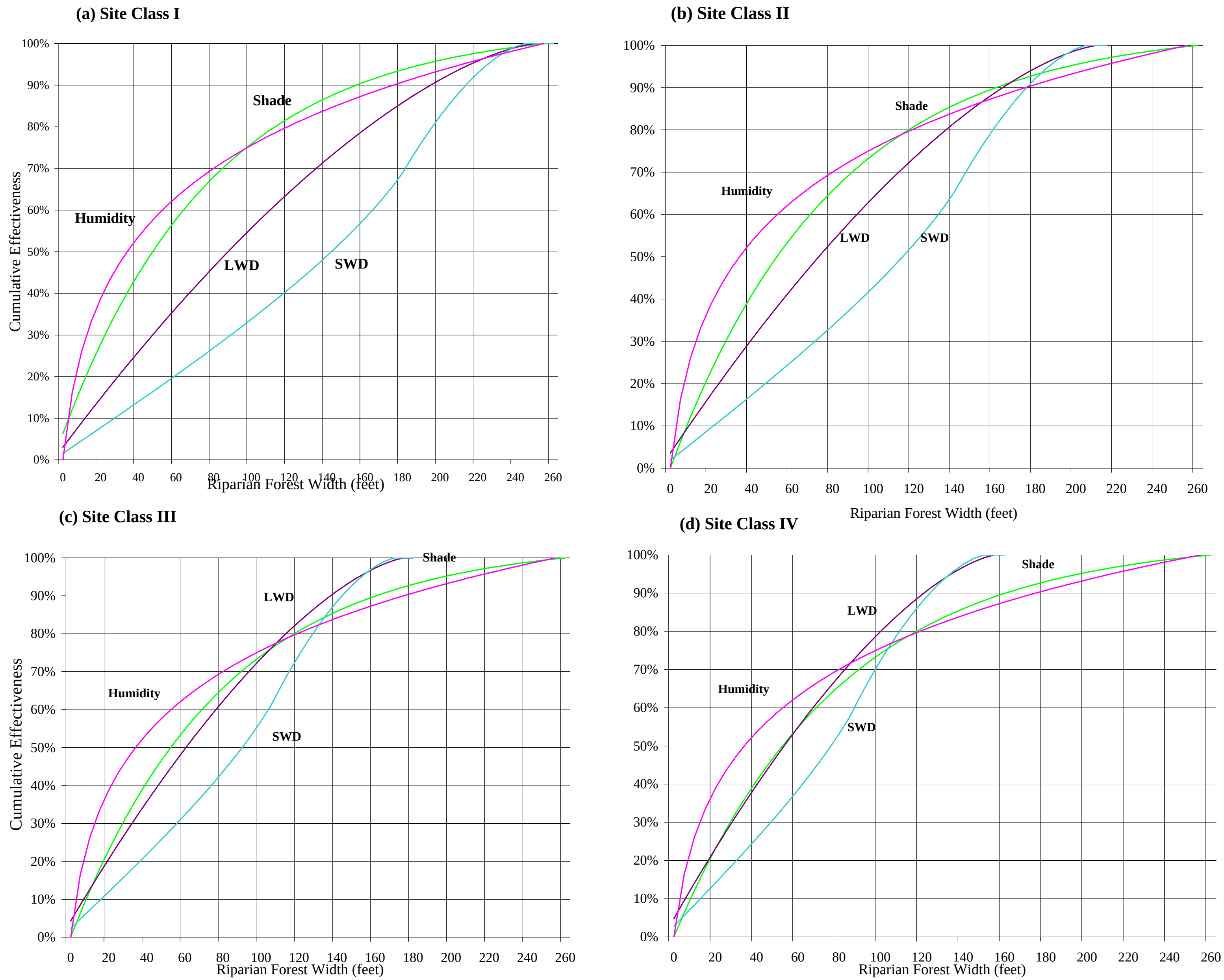


Figure 8. Cumulative benefits of unmanaged riparian forests to streams as a function of distance from stream. (a) Productivity Site Class I forests, (b) Site Class II, (c) Site Class III, (d) Site Class IV. Curves based on McDade et al 1990, Brosofske et al. 1997 and Pollock et al. (in preparation). Site class tree heights are from McArdle et al. 1961, and are for 300 year old Douglas fir forests.

Since all of these materials are important to stream ecosystems, until a quantitative relationship can be developed between the effects of providing less than 100% of these functions and the risk to salmonid populations, riparian forests should be fully functional in terms of being able to provide organic material to streams. Therefore, in the absence of risk analysis, this places a lower limit on riparian buffer widths of between 105-250 feet for Douglas fir forests and 50-250 feet for eastside Ponderosa pine forests, depending on the site potential (McArdle et al. 1930 (Revised 1949), Meyer 1938, see discussion of site potential in Appendix A).

The size of these proposed interim buffer widths may also change if it is determined that providing less than 100% functional riparian forests presents an insignificant risk to salmonid populations. However, if riparian buffers are going to provide less than 100% functional, the relationship between riparian functionality and risk to salmonid populations needs to be quantified, and criteria for determining an acceptable level of risk to salmonid populations needs to be specified.

Most watersheds in Washington have or will soon have one or more salmonid species that are listed under the Endangered Species Act. An acceptable level of risk to endangered species under the Endangered Species Act is a 5% probability of extinction within the next 100 years (Goodman 1996). We assume that 100% functional riparian buffers represents our best current estimate of the narrowest buffer widths that will not appreciably increase the probability of salmonid extinction. If buffer widths are going to be any narrower, they still need to be wide enough to ensure a 95% probability that the species of concern will survive for another 100 years. Determining the size of buffer widths that are not close to fully functional but still provide a 95% probability of success, requires some sort of quantitative risk assessment (sensu Goodman 1996). Since numerous salmonid species are currently imperiled, it would appear that the scientifically and legally appropriate strategy would be to minimize any risk of extinction until such a quantitative risk assessment is undertaken.

In summary, interim buffer widths of 250 feet are proposed for all perennial streams and a width equal to one site potential tree height on seasonal streams. For Douglas fir forests, site potentials range from 105-250 feet, while for eastside Ponderosa pine forests, site potentials range from 50-250 feet (McArdle et al. 1930 (Revised 1949), Meyer 1938, see discussion of site potential in Appendix A). These buffers are intended to ensure that riparian forests return to as close to 100% functionality over the long-term as is reasonably possible, and that the future condition of riparian forests does not contribute significantly to the loss of salmonid populations. These buffer widths are based on the best, currently available scientific information. As more data becomes available, or if a quantitative risk analysis is undertaken, the widths of these buffers will likely change. In particular, we anticipate that on smaller streams, narrower buffers will, in many instances, still provide close to 100% riparian function (once they recover from their current degraded state). While these proposed buffer widths will ultimately minimize the negative affects of riparian conditions on salmonid populations, the continued existence of salmonids in forested watersheds is also dependent on adequate protection elsewhere. In particular, forest practice activities on unstable slopes needs to be minimized, and problems resulting from extensive logging road networks still needs to be addressed.

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Appendix A. Site potential tree height defined and explained

The site potential tree height (SPTH) is the average maximum height to which dominant trees will grow at a particular site if left undisturbed, which is approximated by the tree height at 200-500 years (Sedell et al. 1993) (Figure 9). That is, SPTH is the average height trees will grow at a given site if they do not die prematurely. The potential height of a tree is determined by environmental factors such as climate, soils and hydrology. Site potential is correlated with soil type, and soil maps are often used to estimate the site potential of a particular location.

Forest land is divided into different site classes, often based on the height of the forest at 100 years of age (e.g. McArdle et al. 1930 (Revised 1949), Meyer 1938). Site class I forests are the most productive areas and 100 year old Douglas fir forests are generally between 190-210 feet tall. In contrast, site class V forests are the least productive, and 100 year Douglas fir forests are only 80-90 feet tall. More typical values for 100 year old trees in lower elevation forests in Washington state are in the range of 150-170 feet, with the average probably around 150 feet.

Related to SPTH are site index curves or tables, which describe the relationship between the height and age of a forest for a given site class. Typically, site index curves (or tables) are used to estimate the probable height (or basal area or volume) of a forest to determine when it is appropriate to harvest or thin. The term site index followed by a number, refers to the height of a tree at a particular age. For example, one might refer to typical Douglas fir forests as having a 100 year site index of 150. This refers to forests that are 150 feet tall when they are 100 years old. Confusing the matter somewhat, foresters also refer to a stand with a 100 year site index of 150 feet as having a SPTH of 150 feet, or a 100 year SPTH of 150 feet. For clarity, we always follow the abbreviation SPTH with a subscript specifying the age that we are referring to (e.g. $SPTH_{300}$ for the height of a 300-year old forest).

Most site index tables are based on the height of 100 year old stands. While basing site index tables on 100 year old (or sometimes 50 year old) stands may make sense from a silvicultural perspective, there is no ecological significance to the height of a 100 year old stand. The arbitrary nature of using the height of a forest at 100 years to indicate site productivity can be readily seen from Figure 9, which shows the relationship between tree age and tree height for riparian forests in the Mt. Baker-Snoqualmie National Forest. This graph shows that trees (mostly Douglas fir) grow rapidly in height during the first 150 years, then slowly taper off for the next 150 years and finally after 300-400 years reach their maximum height. The height of a 100 year old stand is just an arbitrary point on the curve that has no real ecological importance, it is just a matter of silviculturist convention. The most ecologically meaningful point on the curve is the age at which a stand first reaches its maximum height, or site potential, since this represents the point in time at which a riparian forest can become fully functional.

For purposes of consistency, we use the standard site index curves of McArdle et al. (1930) for Pacific Northwest Douglas fir forests and those of Meyer (1938) for Ponderosa pine forests to estimate tree heights, and a site index of 300 years to represent the full site potential. While theoretically, a tree will continue to grow in height indefinitely, the average stand height generally quits increasing after three to four centuries because the tops of tall trees often break off during windstorms, thus offsetting height increases due to growth. This phenomenon is reflected in Figure 9, which demonstrates that on average there is little increase in stand height after 300-400 years. Thus, we use the site index curves of McArdle et al. (1930) and Meyer (1938) to estimate the potential height a forest will grow if left unharvested for 300 years for a given site class, and assume that this represents the full site potential.

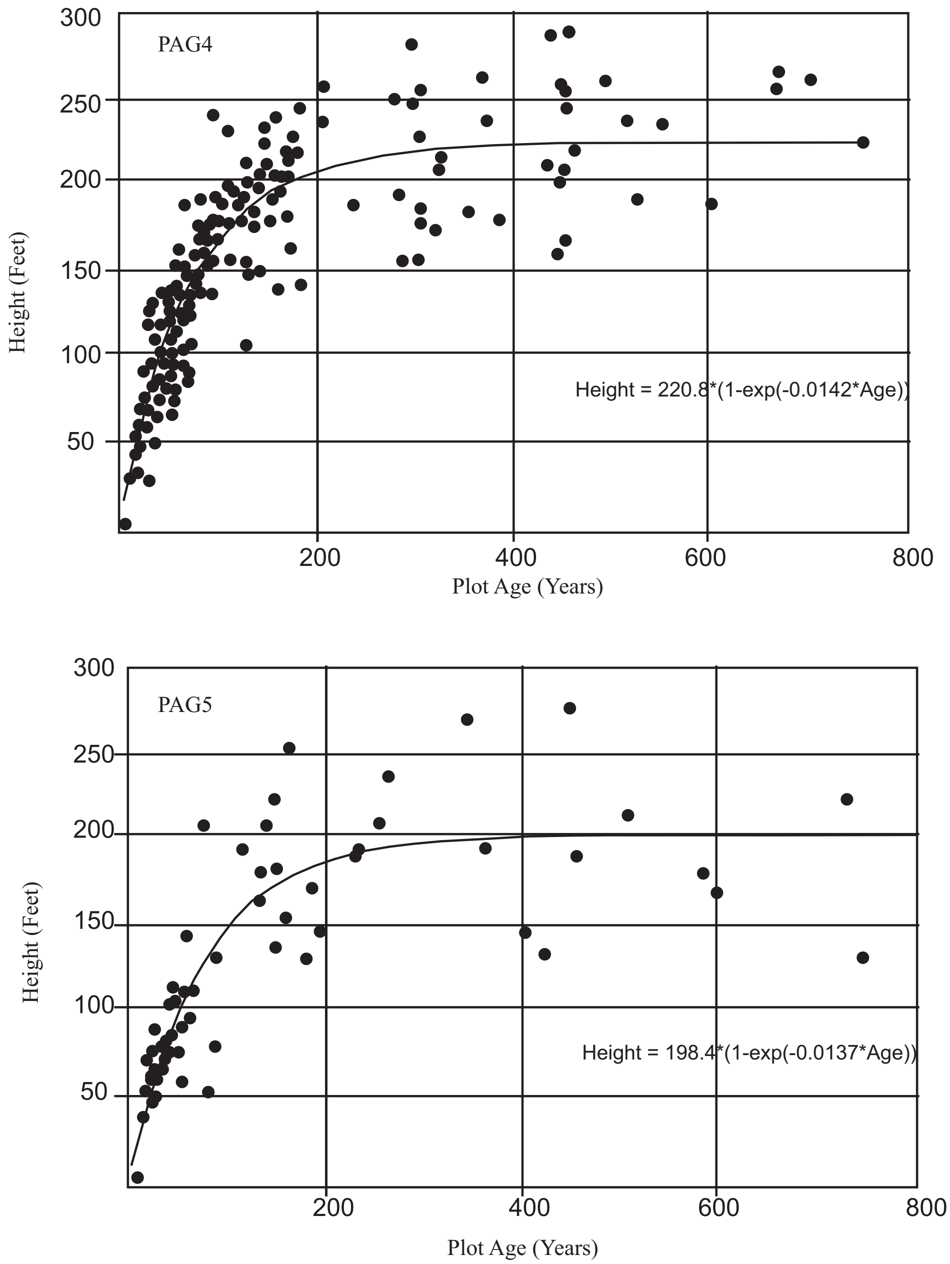

Figure 9. Height v. age for two common riparian plant associations groups (PAGs) for the Mt. Baker-Snoqualmie National Forest, Washington. PAG 4 has a site potential tree height (SPTH) of 218 feet. Site class is approximately equal to a 100 year site index of 170 feet (based McArdle et al. (1961)). PAG 5 has a SPTH of 198 feet. Site class is approximately equal to a 100 year site index of 150 feet. All species are represented (mostly Douglas' fir) and some heights represent damaged tops. Each data point represents one Ecoplot. Data collected between 1979 and 1995. Source: Unpublished Data, Mt. Baker-Snoqualmie National Forest.