APPENDIX B

AQUATIC ECOSYSTEM EVALUATION CRITERIA AND LITERATURE REVIEW

Introduction

This appendix provides a review of the literature on the effects of forest management activities on the environmental components associated with aquatic ecosystems. This review is immediately followed by a discussion of the evaluation criteria used to analyze each environmental component. The topics for which evaluation criteria are considered are listed below under each environmental component:

- Sediment
  - road surface erosion
  - road-related landslides
  - timber harvest-related hillslope erosion
  - timber harvest-related landslides
  - timber harvest effects on bank stability
- Hydrology
  - water yield
  - low flows
  - peak flows
- Riparian Habitat
  - LWD Recruitment
  - Stream shade
  - Leaf and needle litter recruitment
  - Microclimate
- Water Quality
  - Temperature
  - Sediment
  - pesticides

A brief summary of the literature and the evaluation criteria are provided in the EIS text. References are provided in the main EIS reference list (Chapter 4) only so they do not have to be repeated here.

Sediment

Road Surface Erosion

Literature Review

Surface Erosion
Road-related erosion is a function of sediment available for movement and the power of water to move it. Road construction, use, maintenance, abandonment, and drainage all play important roles in the production and delivery of sediment. Surface erosion tends to be a chronic source of fine sediment from roads. Appendix F on forest roads evaluates the specific best management practices (BMPs) of each alternative and should be consulted for further details.

Road Construction
Surface-related erosion from roads appears to be highest in the first few years after construction. Landslide-related erosion and sediment production could occur many years later and is highly episodic (IMST, 1999). Surface erosion from road construction occurs because of exposed soils. The resultant development of gullies on cut slopes and in ditches is a major factor associated with chronic sediment delivery from road prism erosion. Erosion control BMPs can reduce the risk of sediment production from road construction. However, a recent TFW report on the effectiveness of forest road BMPs on state and private forest lands for existing Forest Practices Rules, concluded that the BMPs for road construction (e.g., road crossing structures [culverts] and cut slope construction) were ineffective at preventing chronic sediment delivery for roads near streams (Rashin et al., 1999). Cut and fill slope construction and maintenance BMPs were determined to be partially effective to ineffective at the majority of the sites studied.

Road Use
Several studies have tried to quantify the relationship between road use, road type, and sediment production. In the Clearwater Basin in northwestern Washington, Reid (1981) calculated that road surface erosion contributed 75 percent of total sediments and 82 percent of the total sediments that were smaller than 2 mm. In addition, sediment concentrations increased with culvert discharge for active roads, but showed no increase for abandoned roads; and sediment concentration was higher during periods of heavy road use compared to periods of light road use (Reid and Dunne, 1984). Sediment yield increased with the amount of rainfall in a storm and was higher for roads with a higher intensity of use. Bilby et al. (1989), found that the smallest sizes of sediment (<0.004 mm) comprised about 80 percent of the sediment yield on two mainline road sites and two spur road sites from gravel road surface in southwestern Washington. Burroughs and King (1989) findings for forest roads in Idaho include: (1) heavy traffic on unsurfaced rutted roads doubled sediment production; (2) 6 inches of crushed rock reduced sediment production by 70 percent and by 84 percent when combined with grass at the margins of the travel way; (3) asphalt and road oils reduced sediment production by 97 and 85 percent, respectively; (4) sediment production from unconsolidated fill slopes was high, but decreased exponentially with time. A recent study of forest roads in Oregon (Luce and Black, 1999) concluded:

- Sediment production was a function of road segment length and slope;
- Rocked roads on silty clay loam produced nine times as much sediment as rocked road on a gravelly loam;
- Sediment production was not correlated with the height of the cut-slope;
- Road segments devoid of vegetation (cleared from cut-slopes and ditches) produced seven times the sediment than roads with retained vegetation.

Reduced tire pressure has been shown to reduce sediment production on road surfaces made of low quality aggregate (Foltz, 1996; Foltz and Elliot, 1997). The recent study by Rashin et al. (1999), on forest practices BMP effectiveness in Washington, found that the 4 mainline haul roads monitored in their study were a minor source of sediment production from road prism erosion because they tended to be on flatter topography compared to other roads. However, the sample size in their study was low and may not be representative of mainline haul roads throughout Washington.

Road Drainage
Road drainage refers to the movement of water from road surfaces. Water draining from roads may enter the stream channel or be drained onto slopes. Road drainage issues include: (1) spacing between cross-road drainage culverts; (2) the effectiveness with which these culverts operate; (3) the location at which discharge from a culvert appears; and (4) the management of the outfall from road drainage culverts onto slopes. The drainage from roads directly affects stream channels and their beneficial uses, such as aquatic habitat and water quality.

Several studies on streams west of the Cascade Range (Reid and Dunne, 1984; Bilby et al., 1989; Wemple et al., 1996) have shown that 33 to 75 percent of all drainage from roads enters the channel network. In an unpublished study of road surface drainage by Irvin and Sullivan (cited in Duncan et al., 1987) 20 percent of the road runoff discharged onto the forest floor, while 80 percent emptied directly into the channel network in three large watersheds in western Washington and Oregon. IMST (1999) concluded that reducing the amount of road drainage water that flows into channels can reduce sediment delivery to streams. However, it may not be possible to reduce the number of drainage points into streams, but rather the amount of water that enters the stream can be reduced. The study by Rashin and et al. (1999) concluded that road drainage design BMPs were mostly effective in reducing sediment delivery to streams.

When road drainage is diverted from streams to hillslopes, a potential for problems can be created if drainage onto unstable slopes occurs. This could increase the risk of mass wasting as documented by Montgomery (1994).

Road Maintenance
Road maintenance is an important issue regarding sediment production. Road maintenance is intended to provide a smooth running surface, minimize wear on vehicles, protect road surfacing material, and limit sediment delivery to the stream system. Maintenance of cut and fill slopes, ditches, and removal of slide material on roads can reduce the delivery of sediment to streams. However, maintenance can also be a source of sediment depending upon the maintenance practices.

Road Abandonment

Road abandonment can reduce both chronic and episodic sediment problems from roads. Abandonment varies from simply not using a road to removing stream crossings and the road bed to match the original topography (see Appendix F for a more thorough discussion). IMST (1999) suggested that road abandonment strategies at any given location, and the prioritization of general areas in which abandonment strategies should be implemented, are needed to reduce road-related sediment problems. Abandonment in the current Forest Practices Rules is defined as roads exempt from maintenance after several conditions are met such that the road is left in a condition to reduce and control erosion (see Forest Practice Rules WAC 222-24-050 (5).

Evaluation Criteria

Road surface erosion is affected by road construction methods, road use, road maintenance, road abandonment, and drainage. The criterion for evaluating this chronic source of sediment is a qualitative assessment of how well the FPRs under each alternative that are pertinent to road management (i.e., planning, construction, use, maintenance, drainage, and abandonment) will control road-related sediment production and delivery to streams.

Road-Related Landslides

Literature Review

A landslide is the mass movement of soil, rock, and debris downslope and occurs most frequently after intense winter rains. Landslides are episodic sources of fine and course sediment. Landslide activity can be greatly accelerated by road construction and maintenance practices. On a unit area basis, roads have the greatest effect on slope stability of all activities on forestlands (Sidle et al., 1985). Megahan and Kidd (1972) found 70 percent of accelerated sediment production in an Idaho Batholith study site was associated with road-related landslides. It must be noted this study was prior to the implementation of BMPs with regard to road construction. Piehl et al. (1988) found that 72 percent of the total erosion associated with culvert outlets occurred from only two landslides at culvert outlets.

The location of landslide initiation in relation to the road prism has a significant influence on potential sediment delivery to streams. Landslides affecting the cut slope portion of the road are typically deposited in the road. While surface runoff may erode the deposits of cut slope landslides, the landslide deposit may also divert surface waters away from the designed drainage structures or divert water onto fill-slopes. Fill-slope failures are more likely to become debris flows, increasing in size and then entering intermittent and perennial channels (ODF, 1999).

A study in the Deschutes River basin in Washington concluded that two-thirds of the landslides were road-related and predominantly occurred on roads that were older than 16 years (Toth, 1991). The problems include construction-related problems, such as inadequate drainage, sidecast or fill on steep slopes, poor compaction of fill, backslope steepness, inadequate culvert sizing, and poor culvert installation. Maintenance related problems include debris in ditches, poor grading, and blocked culverts (Toth, 1991). In addition, ODF surveys found that many landslides are the result of roads not constructed to current standards (prior to Forest Practices Rules in Oregon) (IMST, 1999).

A recent ODF study (Robison et al., 1999) on management-related landslides following the 1996 storms in western Oregon concluded:

- Landslides associated with forest roads made up a smaller proportion of the total landslides in the ODF study than road-associated landslides in most previous studies.
- Road-associated landslides were four times larger than those not associated with roads.
- Landslides that delivered sediment to stream channels rarely occurred on roads crossing slopes of less than 50 percent, especially when these roads had well spaced drainage systems and fills of minimal depth.
- Road fill placed on steep slopes creates an increased landslide hazard even where no drainage water is directed to those fills.
- Road drainage waters directed onto very steep slopes creates an increased landslide hazard even when no road fill is placed on those very steep slopes.

Road-related landslides that deliver to streams can become debris torrents and impact stream channels. The impacts to channels are discussed in Timber-Harvest-related Landslides

Orphan Roads
Orphan roads are roads constructed prior to 1974 that have not been used for forest practices since then. The mileage of orphan roads in the state are unknown; however, the associated hazards have been identified. The concern with orphan roads is the potential for failure and initiation of debris flows and torrents. The roads are not maintained and many were constructed with little consideration for public resource and channel impacts. Many roads prior to 1974 were constructed on unstable slopes, in riparian areas, and with substandard drainage systems. Toth (1991) found mass wasting associated with roads was greater with roads greater than 16 years old. The DNR conducted one study of orphan roads to identify the mass wasting hazard of these roads using a hazard ranking index and found several areas within the study area where orphan roads with a high failure potential may impact public resources (Brunengo and Bernath, 1990).

Evaluation Criteria

The potential for road-related landslides depends on both the location of roads in relation to unstable areas and on how the roads are designed, built, and maintained. Therefore, the evaluation criteria for this episodic source of sediment impacts are: (1) the degree to which unstable slopes would be avoided under each alternative; and (2) the degree of protection from road-related landslides provided by the Forest Practices Rules.

Hillslope Erosion

Literature Review

Timber harvest activities often alter watershed conditions by changing the quantity and size distribution of sediment that reaches streams potentially causing channel instability, pool filling by coarse sediment, or introduction of fine sediment to spawning gravels. Factors influencing the delivery of excessive sediment to a stream are discussed in Sections 3.2 and 3.4.

Timber harvest activities such as ground skidding or cable yarding can cause some degree of soil disturbance. Typically, ground-based systems compact and disturb more soils than non ground-based harvest systems (Graham et al., 1990). The harvest systems most likely to cause greater levels of disturbance (from greatest to lowest) are ground-based systems, cable yarding, and aerial systems (Beschta, 1995). For ground-based systems that can operate on slopes up to 40 percent, Hauge et al. (1979) showed a definite relationship between area disturbed and the occurrence of rill and gully erosion. The use of ground-based systems on gentler terrain reduces the potential for erosion. Clearcuts tend to create the greatest amount of soil disturbance, because of the yarding systems they typically use (Hermann, 1978); however, felling, yarding, and skid trails in partial cuts can also cause ground disturbance and compaction. Cromack et al. (1978) found levels of soil disturbance in clearcut and partial cut areas to be comparable because of the need for equivalent access through a harvest unit. Accelerated rates of erosion are generally not prolonged for more than several years as areas revegetate (Beschta et al., 1995).

Riparian protection measures should also include practices for minimizing sediment contributions from outside the riparian area. Timber harvest activities can disturb the upper soil layers, exposing the subsoil to erosion. An important study of sediment delivery from timber harvest was conducted by Rashin et al. (1999) between 1992 and 1995. This study evaluated specific harvest practices on state and private lands across Washington. Harvest sites were evaluated for soil exposed and sediment delivered, and were categorized by a number of parameters, including harvest method and distance from streams. They found that in areas where there were no buffers, BMPs for timber harvest were not effective in preventing soil disturbance nor preventing sediment from reaching streams. They also found that no-harvest buffers of at least 30 feet were effective in filtering sediment, although they caution use of these results because of low precipitation and storm events during the study period.

Overland transport of fine sediment can be significantly reduced by streamside buffer strips. The ability of riparian buffer strips to control sediment inputs from surface erosion depends on several site characteristics, including the presence of vegetation or organic litter, slope, soil type, and drainage characteristics. These factors influence the ability of buffer strips to trap sediment by controlling the infiltration rate of water and the velocity of overland flow.

Evaluation Criteria

The ability of buffer strips to capture fine sediment is largely dependent on their width, slope, and the management practices within the buffer strip. Buffer-strip width is the most common parameter used for evaluating the ability of a management option to avoid excessive fine sediment delivery to streams. Recommended buffer widths for sediment removal vary widely, ranging from about 10 feet for removing coarse fractions (sand) to 400 feet for fine fractions (clay). Studies of forested watersheds often recommend buffers of approximately 100 feet for this purpose (Johnson and Ryba, 1992). Spence et al. (1996) also reviewed the literature on buffer widths for sediment filtration. Although Spence and his colleagues gave no definitive width, they concluded that on gentle slopes 100 feet may be sufficient, while on steep slopes 300 feet may be necessary for sediment filtration. The FEMAT (1993) buffer width for sediment filtration is one SPTH, or approximately 170 feet in western Washington. Note that for management purposes, an absolute width should be used, rather than SPTH, since sediment movement is not related to the latter. Rashin et al. (1999) recommend a

33 foot buffer as effective at reducing timber harvest–related surface erosion from entering the drainage network.

Most of the research conducted to date has assessed buffer effectiveness where there is no management activity within the buffer zone; in other words, only no-harvest buffers have been examined. Notably, Spence and others (1996) recommend that activities which disturb downed wood and ground cover within the riparian buffer zone should be avoided. It can be assumed that activities within the riparian zone that disturb or compact soils, destroy organic litter, and remove large downed wood can reduce the effectiveness of the riparian buffers as a sediment filter, by some unknown amount. Because sediment filtration relies mostly on ground cover, a buffer zone with management activity may recover its sediment filtration ability when the ground cover has recovered. The recovery period would involve many different site-specific variables, such as soil moisture, available light, logging equipment used, yarding practices, and so forth. It is very difficult to assess recovery periods on such a large and diverse landscape as is considered in this EIS. Therefore, while it can be assumed the sediment filtration would be regained after some recovery period, for the purpose of this analysis, and for ease of comparison of the alternatives, a "snapshot" of the buffer is assumed, taken immediately after harvest, to assess the effects. Similarly, because the sideslope of riparian zones is highly variable, the effect of sideslope on sediment filtration is not considered here. This approach is reasonable because the same approach is used for each alternative. Note that because prescribed burns are not common in Washington State, the effects of prescribed burns on buffer strip filtration are not a significant consideration.

To assess the effectiveness of buffer strip width and management practice on sediment filtration, a numerical ranking system was developed based on previous studies of timber harvest and landscape effects. An Equivalent Buffer Area Index (EBAI) for sediment was devised as a crude assessment of risk for streams in relation to management activities. It is similar in concept to the equivalent road area (ERA) analysis of McGurk and Fong (1995) and the non-point source risk assessment of Lull et al. (1995). However, while those studies developed a method to assess sediment contribution from management activities, the EBAI is a relative measure of the protection of streams from fine sediment derived from hillslope erosion and from road surface erosion.

It was necessary to develop the EBAI because studies in the literature typically evaluate buffer widths based on "no harvest," or preservation of mature forest with no disturbance. New management strategies include riparian areas that are divided into zones with different levels of timber harvest activities and thus are not directly comparable to the buffers in the literature. For more detail on the development of the EBAI, see Appendix D. When evaluating the alternatives, the width of the buffers is compared to the typical recommended buffer width of 30 feet, then the activity allowed in the RMZ is evaluated by using the results of the EBAI and by assessing BMPs within the RMZ.

Landslides Related to Timber Harvest

Literature Review

Mass Wasting
A landslide is the mass movement of soil, rock, and debris downslope and occurs most frequently after intense winter rains. Landslides tend to be the dominant erosion mechanism in areas with steep

slopes throughout the Pacific Northwest (Swanson et al., 1987). Landslides are episodic sources of coarse and fine sediment to the drainage network of a watershed. Generally, less than 2 percent of the land is affected by landslides at any given time (Ketcheson and Froelich, 1978; Ice, 1985). Debris slides are the most common landslides on steep forestlands. Major storms increase the rate and intensity of landslides.

Landslides can turn into debris flows (approximately 80 percent solid and 20 percent water) or hyperconcentrated floods, depending on site characteristics and conditions at the time. Debris flows usually transport more material than the initiating slide, due to the scouring action of the debris on the slope or in the channel. Debris flows stop moving when the slope gradient of the channel decreases or when the flow encounters a sharp bend in the channel.

In relation to management-related sediment production, roads contribute more sediment volume via mass wasting than harvest units (Gresswell et al., 1979). Swanson et al. (1987) found the frequency of landslides in the central/western Oregon Cascades was 1.9 to 4 times greater in clear-cuts than unharvested areas. It should be noted that many of these studies were conducted prior to the many revisions in forest practice rules. Mass wasting is significant due to the prevalence, damage caused, and associated costs. On-site damages include loss of timber stand, road blockage or failure, and loss of soil and associated productivity. The most significant off-site damage is the sedimentation or scouring of stream channels and its adverse effects for beneficial uses (water quality and aquatic habitat) and damage to off-site property/buildings and people.

Initiation of Landslides (on site)
Vegetation plays a critical hydrologic and mechanical role on maintenance of slope stability (Greenway, 1987). Hydrologic effects of vegetation on the hillslopes include interception, evapotranspiration, and water routing. Forest practices may alter both physical and biological (vegetative) slope properties that influence slope stability, particularly the occurrence of shallow, rapid landslides. Physical alterations can include slope steepening, slope-water effects, and changes in soil strength. Most physical alterations are a result of roads and skid roads. On a unit-area basis, roads have the greatest effect on slope stability of all activities on forest lands (Sidle et al., 1985).

Mechanical effects of vegetation on slope stability include root reinforcement, buttressing and arching (i.e., trees at base of potential slide act like piles and stabilize the slope), surcharge loading (i.e., weight of trees/logs), windthrow (i.e., soils are displaced and oversteepened by trees blown down by wind), and reduction in slope strength by wedging and loosening soil by roots (ODF, 1999). Sidle et al. (1985) summarized several studies (Swanston, 1970, 1974; O'loughlin, 1974; Ziemer and Swanston, 1977; Burroughs and Thomas, 1977; Gray and Megahan, 1981; Ziemer, 1982) that indicate the stability of slopes depend partly on reinforcement from tree roots, especially when soils are partly or completely saturated. After forest removal, the gradual decay of tree roots often may weaken the soil structure, but the slope will only fail if it was marginally stable to begin with. A recent report prepared by ODF (1999) questions root strength as the primary mechanism affecting slope stability on forestland citing problematic assumptions in landslide analyses, inconsistencies in obtaining representative forest samples, difficult testing procedures, and soil mechanics incompatibilities.

Off-site Impacts of Landslides-Debris Torrents
The effects of landslides on channels are very different than the initial effects of movement. Landslides initiate most debris flows and torrents (Swanson and Lienkaemper, 1978). Debris torrents contain significant amounts of wood and can travel varying distances, which can result in variable degrees of impact depending upon process channel gradient, confinement, layout of the channel network, and other characteristics (Fannin and Rollerson, 1993). Debris flows tend to deposit sediment and debris where channel gradient is less than 6 percent and at tributary junctions where the junction angle is greater than 70 percent (Benda and Cundy, 1990).

Landslides that reach stream channels can have the greatest impact on first to third order channels (e.g., the small streams nearest to the watershed divide) (Swanson et al., 1987). A study of 1996 landslides in Oregon (Robison et al., 1999) found that 40 to 80 percent of the channel network experienced severe impacts from debris torrents in areas that experienced the greatest storm damage to fish-bearing streams. Small streams in a watershed that are severely impacted by landslides also play important roles in supplying wood, sediment, and cold water to fish-bearing streams. Past studies have documented channel changes for small and large streams resulting from high water and landslide activity including channel scour and fill, widening, changes in channel longitudinal profile, and decreases in ecological stability (Lyons and Beschta, 1983; Kaufmann, 1987; Lamberti et al., 1991; Reeves et al., 1995).

An ODF study (1999) made the following conclusions in relation to stream channel impacts from landslides.

- Channel impacts were not directly related to the number of landslides that occurred on the landscape. Large landslides originating above small channel junction angles and steep channel gradient slopes resulted in the greatest channel impacts.
- Debris torrents reduce stream shading especially when they travel through younger stands.
- Debris torrents have only a minor effect on active channel width.
- Less than 10 percent of the typical debris flows from forested slopes traveled further than the Benda-Cundy (1990) model predicts. In determining landslide run-out-distance, channel junction angles and gradient are the primary factors, while landslide volume and composition along riparian area along torrent-prone channels may be important secondary factors.

Channelized debris flows are frequently initiated in headwater areas (e.g., bedrock hollows at the heads of Type 5 streams) (Benda and Dunne, 1985), while dam-break floods can be initiated within streams with relatively gentle gradients (< 20 percent) and often involve dams comprising organic and other debris that has been deposited by upstream debris flows. Once a dam-break flood has begun, it will move downstream unless either of the following happens: the flood wave encounters a predominantly mature coniferous riparian zone with conifers near the channel, or the channel slope decreases to approximately 2 degrees (Coho and Burges, 1994). Similarly, channelized debris flows, once initiated, may scour channels for several miles until reaching depositional environments that include reduced channel gradients, tributary junctions, or increased channel and valley floor widths (Benda, 1985).

Erosion/scour by channelized debris flows usually occurs in first and second-order (Type 4 and 5) channels steeper than 18 percent (Benda and Dunne, 1987). This differs from dam-break floods, which most often scour Type 3 and 4 streams (65 percent and 20 percent, respectively, of cumulative length in study by Coho and Burges, 1994) and have most severe impacts in valleys of widths of at least 20 meters and channel gradients no greater than 7 percent (Johnson, 1991).

Johnson (1991) found that the width of damage to riparian stands ranged from less than 33 feet to 197 feet in a detailed study of four dam-break floods. The average width of 52 measured cross-sections was 75 feet. The most extensive loss of riparian vegetation occurred where valleys widen over 66 feet and slope decreases to less than 7 percent.

Furthermore, the effects of debris avalanches and debris torrents originating from harvested areas may be more damaging than such landslides originating from mature forest. Reynolds and Paulson (1997) documented that the runout along stream channels of debris avalanches which originated in harvested areas was twice as great as those landslides originating in mature forest.

Evaluation Criteria

Mass wasting related to timber harvest is most likely to occur on steep slopes and specific landforms that are highly susceptible to mass failure. The initiation from management activities can occur near streams within riparian areas and upslope areas. The evaluation criteria for harvest-related landslides is the degree of protection provided to unstable areas by Forest Practices Rules. These criteria include protection of unstable slopes upslope from RMZs that may buffer upslope landslides and landslides that may occur in RMZs.

Timber Harvest Effects on Bank Stability

Literature Review

The roots of riparian vegetation help bind soil together, which make stream banks less susceptible to erosion. The stability of stream banks is largely determined by the size, type and cohesiveness of bank material, vegetation cover, and the amount of bedload carried by the channel (Sullivan et al., 1987). Banks formed of noncohesive alluvium that are sparsely vegetated tend to form wide channels (Sullivan et al., 1987). Banks that are composed of more cohesive material tend to form deep, narrow channels (Sullivan et al., 1987). Riparian vegetation of various types and growth forms plays an important role in stabilizing stream banks during the erosive forces of high flows. The belowground root systems serve to bind stream-adjacent sediments, which in turn, influences bank characteristics and the lateral stability of the channel (Beschta, 1995).

Riparian vegetation can also provide hydraulic roughness elements that dissipate stream energy during high or overbank flows, which further reduces bank erosion. In most cases, vegetation immediately adjacent to a stream channel is most important in maintaining bank integrity (FEMAT, 1993); however, in wide valleys with shifting stream channels, vegetation throughout the floodplain or channel migration zone (CMZ) may be important over longer time periods. Lateral river migration through unvegetated floodplains can be twice as rapid as migration through forested floodplains. During overbank flooding, above-ground components of riparian forests provide hydraulic roughness and can exert considerable resistance to flow. This resistance not only reduces the erosive power of the water but typically causes the deposition of sediments (Beschta, 1995).

Consideration should also be given to the composition of riparian species within the area of influence because of differences in the root morphology of conifers, deciduous trees, and shrubs. Specific relationships between root types and bank stabilization have not been documented; however, if the purpose of riparian protection is to maintain or restore natural bank characteristics, then retaining natural species composition is a reasonable target for maintaining bank stabilization function of riparian vegetation (Spence et al., 1996).

Evaluation Criteria

Stream bank erosion can be accelerated by human activity. Important alterations of the system components that typically result from timber harvest activities include the following: (1) removing trees from or near the stream bank; (2) changing the hydrology of the watershed; and (3) increasing the sediment load, which fills pools and contributes to lateral scour by forcing erosive stream flow against the stream bank (Pfankuch, 1975; Cederholm et al., 1978; Chamberlin et al., 1991). This evaluation is based on the widths of the respective RMZs and activities allowed within the RMZ that may affect root strength and thus stream bank integrity. Increases in sediment and changes in hydrology that ultimately can affect bank stability are addressed in Section 3.2 and Section 3.3, respectively.

As described in Section 3.3.2, the roots of riparian vegetation help bind soil together, which makes stream banks less susceptible to erosion. In most cases, vegetation immediately adjacent to a stream channel is most important in maintaining bank integrity (FEMAT, 1993); however, in wide valleys with shifting stream channels, vegetation throughout the floodplain may be important over longer time periods. Although there is limited data quantifying the effective zone of influence relative to root strength, FEMAT (1993) concluded that most of the stabilizing influence of riparian root structure is probably provided by trees within 0.3 SPTH of the stream channel.

Consideration should also be given to the composition of riparian species within the area of influence because of differences in the root morphology of conifers, deciduous trees, and shrubs. Specific relationships between root types and bank stabilization have not been documented; however, if the purpose of riparian protection is to maintain or restore natural bank characteristics, then retaining natural species composition is a reasonable target for maintaining bank stabilization function of riparian vegetation (Spence et al., 1996).

Overall, buffer widths for protecting other riparian functions (e.g., LWD recruitment, shading) are likely adequate to maintain bank stability providing that they are maintaining most of those functions. For this analysis, one-half of a tree crown diameter (which is in the range of 0.3 SPTH) is assumed to be sufficient width for the maintenance of stream bank stability. The contribution of root strength to maintaining streambank integrity also declines at distances greater than one-half a crown diameter (Burroughs and Thomas, 1977; Wu, 1986) The full assumed relationship is displayed in Section 3.4 of the EIS.

Hydrology

Water Yield

Literature Review

Water yield is the amount of water that enters the stream system in a watershed. Various studies (Helvey, 1980; Bosch and Hewlett, 1982; Harr, 1983; Kattlemann et al., 1983; Troendle, 1983; King and Tennyson, 1984; Trimble and Weirich, 1987; Keppeler and Ziemer, 1990) have shown increases in water yields from forest practices, specifically timber harvest, due to the reduction in evapotranspiration.

Evaluation Criteria

An increase in water yield is generally not considered to adversely affect the beneficial uses of the stream system and will not be evaluated; increased water to the aquatic system is usually considered beneficial, unless the added water contributes to slope instability or increases peak flows.

Low Flows

Literature Review

Low flows are often referred to as baseflows, dry-weather flows, and groundwater flows depending on the specific context. On the west side, low flows are most common in the summer and early autumn months until the winter rainy season begins in late autumn. At higher elevations, low flows also occur in mid-winter when water is frozen. On the east side, late summer flows can continually decrease throughout the winter months until snowmelt begins in the spring. Several studies (Rothacther, 1970; Harr et al., 1982) have shown an increase in summer low flows for west side forests after harvest. The effects on snowmelt-dominated systems like the east side can be variable based upon climatic and watershed conditions (Beschta et al., 1995).

Evaluation Criteria

Because an increase in low flows for summer months generally does not adversely affect the beneficial uses of the aquatic system, it will not be evaluated. Small volumetric increases may provide improved habitat conditions (lower stream temperature, increased instream wetted area and volume) and survivability of aquatic species.

Peak Flows

Peak flow is the instantaneous discharge measured in stream channels. Management activities can affect peak flows based upon their site specific effect, elevation location within a watershed and proportion of basin forest that has been altered by activities, such as roads and timber harvest. Changes in peak flows can adversely affect channel conditions, bank stability, fish habitat, water quality, public works, and safety.

Effects of Roads on Peak Flows

Literature Review
Watershed characteristics such as topography, soil, and geology affect the road design and the potential for road failure. Thus, roads may be more concentrated in certain areas because of the engineering constraints with road design and construction in steep areas or unstable slopes. The design, construction, and maintenance of roads interact with these watershed characteristics to determine how roads can alter the hydrologic behavior of a particular basin. The interception and routing of surface runoff and interception of subsurface flow by a road prism can affect the flow patterns of a watershed. A wide range of management factors (e.g., location, excavation method, design standard, surfacing, drainage system, road density, level of use, and maintenance) affect the road associated impact to hydrologic processes. These management factors will be used as evaluation criteria for potential road influences on peak flows.

Roads can act as an extension of the drainage network. Roads intercept groundwater in road cuts, surface flow from small drainages, and direct rainfall (Best et al., 1995; Megahan, 1975). Roads can gather and transmit rainfall faster than the natural landscape, altering flow patterns in the basin (Harr et al., 1975; Harr et al., 1979; Jones and Grant, 1996). The effect of roads on the extension of the drainage network decreases as watershed size increases, and increases in peak flows in small watersheds become alternated and desynchronized as they move downstream (Beven and Wood, 1993). Increases in peak flows though these effects are typically evident only in smaller basins (Ziemer and Lisle, 1997). In a study in western Oregon, roads caused the stream density (length of streams per unit area) to increase 38 percent over the pre-road conditions (Wemple, 1994). Results of studies are mixed, however. In some cases, the effect of roads cannot be differentiated from the effects of timber harvest, because they usually occur simultaneously or in quick succession. In other cases, where timber harvest was effectively factored out, there was no significant change in peak flows (Wright et al., 1990; Ziemer, 1981). Road location within the watershed studied may be a factor; if roads are closer to the main stem, their contribution to hastening subsurface flow concentration would be small. On the other hand, Jones and Grant (1996) showed a significant increase in peak flows after road building. This was later questioned in a paper by Beschta et al. (1997). Harr et al. (1975) also showed an increase in peak flows related to roads.

Evaluation Criteria
Although the results of studies are varied, there is a potential that road drainage may play some role in peak flow events, which would have greater impacts on first and second order drainages (i.e., small watersheds). This potential may be significant in certain basins or storms, and will be evaluated based upon the road management and drainage criteria and potential for decrease (e.g., abandonment) in roads under each alternative.

Effects of Timber Harvest on Peak Flows

Literature Review
The best understood effect of timber harvest is its influence on streamflow relating to altering snow accumulation and melt rate. Increased peak flows can occur in the winter, when a warm wet storm brings rain after a cold storm deposits significant amounts of snow. The snow melts much faster than from warming of the air temperature alone. Many floods in Washington, mostly on the west side of the Cascades, have occurred as a result of rain-on-snow (ROS) events. While ROS events

are a natural occurrence, their effects can be exacerbated when a watershed has a large percent of immature vegetation (Coffin and Harr, 1992; Troendle and Leaf, 1980). The two most important watershed variables that affect ROS processes are elevation and extent of timber harvest.

Timber harvest has the potential to alter snow accumulation and melt rates anywhere. Forest openings are conducive to increased snowpack accumulations: more snow reaches the ground because there is less tree canopy to intercept the snow where it could melt. Once a rainfall occurs, the forest openings are more conducive to higher rates of convection and condensation to the snowpack than the surrounding forest. The combination of greater snow accumulation and increased melt rates can lead to a greater amount of water available at the soil surface in forest opening during a ROS event than occurs in the adjacent forest (Coffin and Harr, 1992). The net result is that increased runoff is expected from clearcuts during ROS events.

The effects of timber harvest and road building on peak flows tend to be more noticeable in small basins than in large basins, because a large percentage of a small basin may be affected at one time. In addition, large basins respond differently to hydrologic effects than small basins. Stormflow response in smaller watersheds is determined more by hillslope parameters (which can be affected by timber harvest) than by channel features (Robinson et al., 1995). The reverse is true for larger watersheds. There is limited evidence that large basins may be affected by timber harvest. In basins up to 230 mi^2, Jones and Grant (1996) observed increased peak flows among small storms (one-year recurrence interval) but speculated that runoff from larger storms would be affected as well. This was later questioned in a paper by Beschta et al. (1997).

Though the effects are usually minor, recovery of peak flows from hydrologic changes in a forest tends to be gradual. In western Oregon, Grant (1994) observed little peak flow recovery after 30 years. In another study, 50 percent hydrologic recovery was achieved in 25 years (Harr et al., 1989).

Evaluation Criteria
Many studies have found a correlation between the hydrologic maturity of basin in the ROS elevation zone and the potential for increased peak flows. The evaluation criteria for harvest-related peak flows is how well the Forest Practices Rules under each alternative reduce the potential for large land areas in watersheds to become hydrologically immature (i.e., little crown closure) to protect from increased peak flows in the ROS zone.

Riparian Habitat

LWD Recruitment

Literature Review

Riparian areas are an important source of LWD that enters, or is recruited to, the stream channel. LWD includes entire trees, rootwads, and larger branches. Numerous studies have shown that LWD is an important component of fish habitat (Swanson et al., 1976; Bisson et al., 1987; and Naiman et al., 1992). Trees that fall into streams are critical for sediment retention (Keller and Swanson, 1979; Sedell et al., 1988), gradient modification (Bilby, 1979), structural diversity (Ralph et al., 1994), nutrient production (Cummins, 1974), and protective cover from predators. LWD also creates storage sites for sediment in all sizes of streams. In small headwater streams, wood controls sediment movement downstream minimizing the risk of debris flows. In larger streams accumulation of sediment behind LWD often provides spawning gravels. LWD plays an important role in stream nutrient dynamics by retaining leaf litter and needles, making these energy supplies available for consumption by aquatic insects that ultimately serve as food for fish.

The large wood recruitment mechanism originates from a variety of processes including tree mortality (toppling), windthrow, undercutting of stream banks, debris avalanches, deep-seated mass soil movements, and redistribution from upstream (Swanson and Lienkamper, 1978).First and second-order headwater streams can also provide wood to larger higher order channels downstream (Potts and Anderson, 1990; Prichard et al., 1998). Two predominant mechanisms have been observed for the movement of LWD between stream types: transport during high flow events and debris torrents, which includes dam-break floods and debris flows (Swanson and Lienkaemper, 1978). However, the former mechanism is more common in third to fifth order streams because much of the wood that falls into streams is too large to float in smaller streams (Swanson and Lienkaemper, 1978). The occurrence of debris flows, although less frequent than the redistribution of LWD from high flows, can introduce large amounts of LWD. One 840-meter debris flow was cited as bringing more than 700 pieces of wood into fish-bearing waters (Lamberti et al. 1991). Additionally, the rate of occurrence of debris flows originating from management actions in forests (albeit, under older less protective rules) occurred at a rate of 450% to 4,100% higher than that of unmanaged forests (Swanson 1976). Morrison (1975) found that forest practices increased debris torrents 880% to 1300% when compared to unmanaged forests. The greatest number of debris flows and dam-break floods are initiated in lower order streams with the highest number of events starting in second order streams. The majority of dam-break floods and debris flows traveled between two and four thousand meters (Coho and Burges 1991). Dam-break floods that begin in low order channels may eventually enter into and impact high order, low gradient channels, and cause significant disturbance to riparian vegetation and aquatic habitat during and after the event (Coho and Burges 1991). "The most obvious schemes for avoiding the destructive forces of organic debris movement are maintaining contiguous riparian zones of mature conifers around low order channels and minimizing deposition of logging slash and debris into those channels" (Coho and Burges 1994).

The potential size distribution of LWD is also an important factor when considering the appropriate activities in buffer strips relative to LWD recruitment. There is a strong relationship between channel width, and the size of LWD that forms a pool (Bilby and Ward, 1989; Bilby and Wasserman; Beechie and Sibley, 1997; Beechie, 1998). LWD that is large enough to form a pool is referred to as "functional LWD." Smaller pieces of LWD can function in smaller streams and larger pieces are needed in larger streams. Although only a certain percentage of functionally- sized LWD may actually create pools, the greater the proportion recruited, the greater the potential for pool formation. Data from western Washington indicates that 3 of 10 pieces of functionally-sized LWD actually create pools (Kennard and Pess in press, Montgomery et al., 1995; Beechie, 1998). Bilby and Wasserman (1989) documented regional differences in piece size that is considered functional in similarly sized system. The piece size considered functional was smaller in eastern Washington streams than in similarly sized systems west of the Cascades. Hydrologic conditions in eastern Washington display smaller extremes than in western Washington.

"Key piece LWD," which is considered a subset of "functional LWD," is considered by some to be a better measure of the important wood recruitment sizes. Key piece size includes pool-forming capacity similar to "functional wood size," but also takes into consideration the effectiveness of the piece in trapping other smaller more mobile pieces of LWD (forming logjams), as well as the long-term stability of pieces. The watershed analysis manual identifies the specific LWD sizes that meet the key-piece definition. These sizes are based on data collected by Fox (unpublished). Similar to Bilby and Ward (1989), Bilby and Wasserman (1989), Beechie and Sibley (1997) and Beechie (1998) minimum LWD size increases with channel width (Washington Forest Practices Board 1995). Key piece size for a stream 40 feet wide is approximately 15 percent larger then functional wood size for the same width stream. For larger streams, the difference can be greater. As a result, RMZs need to ensure not only an appropriate amount or volume of wood, but wood of sufficient size to serve as both functional pieces and key pieces (Murphy, 1995).

The length of time needed for riparian areas to produce LWD after harvest depends upon the size of the stream. Measurable contributions of wood from second-growth riparian areas did not occur until 60 years after harvest for third-order channels on the Olympic Peninsula in Washington (Grette, 1985). Bilby and Wasserman (1989) indicate that it takes longer than 70 years for streamside vegetation to provide stable material to streams wider than 50 feet in southwestern Washington. Murphy and Koski (1989) indicate that post-harvest LWD recruitment levels have relatively long recovery rates of up to 250 years in Southeast Alaska. Therefore, larger streams are likely to be deficient in LWD for a longer period of time after timber harvest than smaller streams (MacDonald et al., 1991).

Evaluation Criteria

As described above, large wood plays an important role in aquatic habitat. The large wood recruitment mechanism originates from a variety of processes including tree mortality (toppling), windthrow, undercutting of stream banks, debris avalanches, deep-seated mass soil movements, and redistribution from upstream (Swanson and Lienkamper, 1978). Riparian forests (or RMZs) are an important source for recruiting almost all instream LWD. Thus the quality of instream habitats is largely determined by how the riparian forests have been managed. Though it is recognized that unstable upslope areas adjacent to headwater streams, are often an important source of LWD (IMST, 1999; Prichard et al., 1998), the focus in this section is on LWD recruitment from riparian areas.

The potential for trees to enter a stream channel from tree mortality, windthrow, and bank undercutting is mainly a function of slope distance from the stream channel in relation to tree height. As a result, the zone of influence for LWD recruitment is determined by the particular stand characteristics rather than an absolute distance from the stream channel or floodplain. Slope and prevailing wind direction are other factors that can affect the amount of LWD recruited to a stream (Spence et al., 1996). To maintain full recruitment potential of LWD to the stream channel, protection of all trees within the zone of influence is required.

FEMAT (1993) concluded that the probability of wood entering the active stream channel from greater than one SPTH is generally low. McDade et al. (1990) estimated that for old-growth conifer forests in Oregon, 50 percent of debris originates within 39 feet of the stream, 85 percent within 100 feet, and 100 percent within 165 to 182 feet. McDade et al. (1990) values for mature forest are 33 feet, 75 feet, and 154 feet, respectively. They also showed that 90 percent of LWD in mature forests originated within 89 feet of the stream channel. Two widely used models of LWD recruitment also assume that large wood from areas outside one SPTH seldom reaches the stream channel (Van Sickle and Gregory, 1990; Robinson and Beschta, 1990). Cederholm (1994) reviewed the literature regarding recommendations of buffer widths for maintaining recruitment of LWD to streams and found most authors recommended buffers of 100 to 200 feet for maintaining this function. A riparian buffer consisting of taller older trees contributes large wood from greater distances than do younger forests with shorter trees (McDade et al., 1990; Van Sickle and Gregory, 1990; Fetherston et al., 1995). In summary, most recent studies suggest buffers approaching one SPTH is sufficient to maintain 100 percent of natural levels of LWD recruitment (Spence et al., 1996) (see figure in Section 3.4 of the EIS).

In addition to the amount of LWD input, the species of LWD contributed is also important. Coniferous LWD significantly outlasts deciduous LWD in the stream system (Harmon et al., 1986; Grette, 1985). Simply setting aside buffers of second-growth hardwoods does not provide optimal LWD input over the short-term because unassisted recovery of these areas to pre-logging coniferous LWD recruitment levels may take over 100 to 200 years.

For evaluating the alternatives, both buffer width, which determines the area from which potential source trees can contribute LWD, and prescriptions, which determine how much of this potential material remains after timber harvest (Murphy, 1995) were considered. An index that incorporates both buffer width and prescription, was used to assist in the evaluation of alternatives by quantitatively measuring the potential of each alternative to provide LWD to streams of varying size (see Appendix D). This index is referred to as the equivalent buffer area index (EBAI) for LWD. This index is based, in part, on the mature conifer curve displayed in Section 3.4 of the EIS and a slightly different curve for old-growth conifers. The old-growth curve has a higher proportion of the total recruitment potential derived from a distance further from the stream. Both curves assume that full recruitment of LWD (by toppling, windthrow, or stream undercutting) occurs if no-harvest riparian buffers of one site-potential tree height are retained. An exception to this may occur in second-growth stands where hardwoods have excluded regeneration of coniferous trees or overstocking of stands have lead to the depletion of large size classes of debris (Spence et al., 1997). As a result, growth rate modeling of tree diameter and age to reach functional and key piece recruitment size, based on different silvicultural prescriptions and different stream sizes was also used when evaluating alternatives (see Appendix D).

Stream Shade

Literature Review

There are several factors that make up the heat balance of water (see Section 3.4), which determine how the temperature of a stream will change as it flows downstream including: air temperature, solar radiation, evaporation, groundwater inflow, convection, conduction, and advection (Brown, 1983, Adams and Sullivan 1989). In addition, seasonal and daily cycles produce a high degree of variability in stream temperature. Other site-specific factors such as latitude, proximity to the ocean, stream order, distance from watershed divides, upslope soil temperatures, elevation, stream geometry and orientation, local topography, riparian tree species, stand age, and stand density have all been correlated with stream temperature changes (Beschta et al., 1987; Sullivan et al., 1990, Brosofske et al. 1997, Lewis et al. 2000). Notably, streams have a natural tendency to warm as they progress from headwaters to the ocean even when complete canopy closure is present (Sullivan et al. 1990). During the summer, when stream temperatures are the highest, the combination of air temperature, direct solar radiation, a decrease in stream flows, and the amount of groundwater inflow have the greatest effect on stream temperature changes downstream (Beschta et al., 1987, Adams and Sullivan 1989). Air temperature is the most important factor, but the relationship between air and water temperature is not linear because evaporation and radiation loss increase at higher water temperatures (Sullivan et al. 1990). Consequently, there is a maximum equilibrium water temperature even if air temperature is increased further (Sullivan et al. 1990). Of the four factors, forest management can have the greatest effect on direct solar radiation. Forest management can also affect soil and air temperature (St-Hilaire et al. 2000, Brosofske et al. 1997, Chen et al. 1995), but the degree to which these factors result in substantial water temperature changes is less well understood.

Water temperature modeling approaches are typically statistical or mechanistic. Statistical models collect data for a variety of variables and then construct a model based upon the best statistical fit, often using multiple regression techniques. These models generally use variables such as elevation combined with regional weather data as surrogates for air temperature. Air temperature is highly correlated with elevation and is described by the environmental temperature lapse rate. The average environmental temperature lapse rate is about 1.0°C for every 300 feet of elevation, but could be somewhat higher or lower depending upon specific site characteristics of location and time. Mechanistic models such as SNTEMP (Theurer, et al. 1986) are based upon an understanding of the physics involved with stream heating. The model developed for the DNR Forest Practices Rules manual, TFWTEMP (Doughty et al. 1993), is primarily a mechanistic model, but uses regional regressions to model air temperature as an input parameter. SNTEMP and TFWTEMP both include parameters for examining the effect of different shade levels that might result from forest practices. St-Hilaire et al. (2000) extended a mechanistic water temperature model, CEQUEAU, by including the effects of soil heating on interflow (horizontal movement of water above the water table) that results from removing upslope canopy cover (i.e., from a clearcut). Their model has been calibrated using existing data, but its predictive ability has not been tested against independent data (St-Hilaire et al. 2000).

Studies of microclimate gradients have demonstrated increases in air and surface soil temperatures (among other factors) as edge effects related to clearcut harvest practices (Chen 1991, Chen et al. 1993, Chen et al. 1995, Brosofske et al. 1997). However, of the three papers only Brosofske et al. (1997) specifically examined microclimates in riparian zones while the three studies by Chen and his colleagues examined microclimate upslope from streams. Notably, Brosofske et al. (1997) observed statistically significant changes between pre- and post-harvest for several microclimate variables (air temperature, surface soil temperature, and relative humidity), but no significant relationship between stream water temperature and stream buffer width was observed except at the site with the smallest (nearly non-existent) buffer. Significant positive linear correlations between water temperature and shallow (4 inches) upslope soil temperatures were documented and Brosofske et al. (1997) suggested the results indicated a causal relationship because their study occurred along 1st order streams that received groundwater that passed through upslope soils. However, the focus of the paper by Brosofske et al. (1997) was not water heating and hydrology; neither interflow or groundwater temperatures were measured. Chen et al. (1995) have demonstrated that increases in air temperatures decline along a gradient from the edge of a clearcut (up to 7.8°C higher) extending 787 feet into the interior of adjacent old growth stand. Similarly, a gradient in surface soil temperature (up to 9.3°C higher at the clearcut edge) was also demonstrated, but the effect was relatively small at distances more than 30 meters from the clearcut.

Overall, the results from Brosofske et al. (1997) and Chen et al. (1995) are provocative and suggest effects that may be important to amphibians and other riparian-dependent species; but the strength of an effect, if any, on water temperature is unclear. The model by St-Hilaire et al. (2000) under assumptions for a severe tropical storm event during August in a small New Brunswick watershed (12,864 acres) predicted small increases in stream heating (<0.5°C) when canopy cover is reduced about 10% over the watershed. Under 50% and 100% canopy removal assumptions, temperatures were predicted to increase 0.9°C and 4.0°C, respectively (St-Hilaire et al. 2000). Consequently, their model suggests that relatively large levels of canopy removal are necessary to cause substantial increases in water temperature. Zwieniecki and Newton (1999) examined the effects of different levels of stream canopy cover and stream geometry characteristics on water temperatures in low elevation streams in Oregon. Their results suggested that water temperatures in streams exiting clearcut harvest units with 28 – 100-foot wide buffers returned (declined) to the expected temperature trendline within 492 feet downstream of the harvest unit after accounting for the natural warming of the stream.

Direct solar energy is the largest component of energy available to warm stream water (Chamberlin et al., 1991). The more riparian trees are removed, the more that shade is reduced and the more energy reaches the stream, resulting in a potential increase in stream temperature. While shade cannot physically cool the stream down, it can prevent further heating of the stream, maintaining the cool water from groundwater inputs or tributaries (Forest Practices Advisory Committee, 1999). Many studies have documented increases in stream temperature due to timber harvest. The degree of impact varies with particular practices and stream characteristics. However, harvesting to the stream bank without leaving trees, has consistently been shown to increase mean, maximum, and diurnal fluctuation of stream temperature (Meehan, 1970; Holtby, 1988). Shade provided by riparian vegetation has been shown to be successful in minimizing or eliminating increases in stream temperature associated with timber harvest (Brazier and Brown, 1973; Lynch et al., 1985).

Evaluation Criteria

As described in the literature review, several site-specific factors including composition of vegetation, stand height, stand density, latitude (which determines solar angle), topography, and orientation of the stream channel play a role in whether riparian habitat provides shade to stream channels. These factors influence how much incident solar radiation reaches the forest canopy and the fraction that passes through to the water surface (Spence et al., 1996). Belt et al.'s (1992) review of a number of studies indicated that any removal of forest canopy within the buffer strip reduces its effectiveness by reducing shade and thereby increasing stream temperatures.

Brazier and Brown (1973) used angular canopy density (ACD) to measure shade. ACD is different than canopy closure in that ACD is the canopy density in the direction from which maximum direct beam sunlight originates. In contrast, canopy closure is the canopy density over the view the sky observed by the measuring device (a spherical densiometer, moosehorn or other device). Brazier and Brown (1973) found that an ACD comparable to old-growth stands (i.e., 80 to 90 percent ACD) could be attained with buffers of approximately 72 feet to 100 feet for coniferous forests in the southern Cascades and Oregon Coast Range. Steinblums et al. (1984) determined that an ACD of approximately 100 percent could be achieved with the use of buffer strips greater than 125 feet. Based primarily on these studies, several authors have concluded that buffers of 100 feet provide adequate shade to stream systems (Murphy, 1995; Johnson and Ryba, 1992). If the buffer is less than 100 feet, or if the buffer is selectively logged, considerations such as species composition, stand age, and vegetation density become important factors (Beschta et al., 1987). Beschta et al. (1987) concluded that 100 feet provides 100 percent of the ACD typically observed in old growth. The generalized curves presented by FEMAT (1993) for forests in the range of the spotted owl suggest that cumulative effectiveness for shading approaches 100 percent at a distance of approximately 0.75 tree heights from the stream channel (see figure in Section 3.4 of the EIS). For a forest with an average site potential tree height of 160 feet, a 100 percent effective buffer for temperature based on the generalized FEMAT curve can be expected to be approximately 120 feet.

For small perennial streams (<5ft width) there is very little published literature. However, small streams are often completely shaded by woody vegetation and therefore have no riparian canopy opening in their undisturbed state. Caldwell et al. (1991) found that 40 percent shade was provided to west side Type 4 streams from brush and slash following timber harvest to the stream bank. Broderson (1973) found that a 50-foot buffer provided 85 percent of the maximum shade for small streams (defined as streams with mean annual discharges of less than 5 cfs).

Small streams are more sensitive to rapid temperature changes because of their lower water volume (Sullivan et al. 1990). In areas where partial or complete exposure of the stream causes increased stream temperatures the rate of shade recovery depends on streamside conditions, vegetation, as well as, stream size (Beschta et al., 1987). Small streams may be quickly overtopped by brush and effectively shaded from solar radiation, while larger streams, which require tall conifers for shade require longer time periods. Reestablishment of canopy cover over streams can range from 5-40 years or more (Gregory and Bisson, 1997).

The sensitivity of streams to changes in water temperature may vary regionally. Such regional differences in temperature sensitivity are due to a number of factors including elevation (Rashin and Graber, 1992), proportion of small nonfish-bearing streams and proximity to the coast (TFW Temperature Work Group, 1990). Because stream temperature decreases with increased elevation, streams at higher elevations are expected to be cooler and the need for shade to maintain water temperature below standards is less than downstream areas. These factors will be discussed under the water quality sections under water temperature. For the purpose of the EIS analysis, the focus is on how the different alternatives RMZs will provide for complete shading of the aquatic system, not taking into consideration whether the criteria will meet or exceed water quality standards.

Acknowledging that there is site-specific variation that determines shade requirements, it was concluded that buffer widths of approximately 0.75 site-potential tree height are needed to provide complete protection of stream shading along most streams. The full curve, identifying the estimated relationship between shade protection and RMZ width, used to evaluate the alternatives is shown in a figure in Section 3.4 of the EIS. However, along small perennial streams (<5 ft wide) a 50-foot RMZ was used as the minimum criteria to provide sufficient shade to maintain stream temperatures since understory trees and shrubs play a much more important role in shading smaller streams. This assumption is supported by Broderson (1973) who found that a 50-foot buffer provided 85 percent of the maximum shade for small streams (defined as streams with mean annual discharges of less than 5 cfs). For seasonal streams that do not flow during the summer stream shade should have minimal to no effect on temperature and therefore will not be considered when evaluating shade requirements.

Leaf and Needle Litter Recruitment

Literature Review

Stream benthic communities (e.g., aquatic insects) are highly dependent on detrital inputs. Detritus is defined as all dead organic carbon, distinguishable from living organic or inorganic carbon (Wetzel, 1975). With respect to stream systems, detritus has two sources: detritus originating within the stream (autochthonous); and detritus originating from outside the stream (allochthonous). The primary form of autochthonous detritus is dead algae and other aquatic plant material. In small forested mountain streams, autochthonous detritus accounts for only a small portion of the total detrital input within the system. Allochthonous detritus is the primary source of detrital input into small and medium size streams through the annual contribution of large amounts of leaves, cones, wood, and dissolved organic matter (Gregory et al., 1991; Richardson, 1992). The importance of this type of detrital input varies among streams, but can provide up to 60 percent of the total energy input into stream communities (Richardson, 1992).

The size and morphology of a stream greatly influences the deposition and processing of organic materials. Litter deposited into small steep-gradient streams in forested areas high in a watershed is often transported rapidly downstream because little of the area is suitable for retention of organic material. Therefore, these small streams are important to the nutrient cycling and productivity of larger streams in lower reaches of the watershed (IMST, 1999).

Evaluation Criteria

Allochthonous detritus enters a stream primarily by direct leaf or debris fall, although organic material may also enter the stream channel by overland flow of water, mass soil movements, or shifting of stream channels. Few studies have been done relating litter contributions to streams as a function of distance from the stream channel; however, it is assumed that most fine organic litter originates within approximately 0.5 tree heights from the channel (FEMAT, 1993). In hardwood woodlands, windborne leaf litter may travel farther from source trees than needles or twigs from coniferous vegetation. In addition, most needles are retained by coniferous species during the fall and winter, while deciduous hardwood species lose all of their leaves. Therefore, riparian buffers in these hardwood stands may need to be wider than in coniferous forests to protect natural levels of organic inputs. However, buffers designed to provide for LWD recruitment generally will provide nearly 100 percent of allochthonous detritus (Spence et al., 1996). Spence et al. (1996) concluded that a buffer width of 0.75 site-potential tree heights is needed to provide for 100 percent of litter inputs.

Forest practices can lead to changes in leaf litter distribution and dynamics in upland areas and riparian areas, which in turn affect availability in streams. Harvest intensity (i.e., proportion of forest canopy removed), and cutting frequency all affect the rate of nutrient removal from the system (Beschta et al., 1995). As a forest ages, litterfall increases with increasing biomass. Therefore, stand age significantly influences detrital input to a stream system. Allochthonous detrital input was estimated to be two times as high in old-growth forests when compared to either 30- or 60-year-old forests (Richardson, 1992) and can be as much as five times as high when compared to a clearcut forest (Bilby and Bisson, 1992). However, reduced levels of allochthonous detrital input into streams due to streamside timber harvest is somewhat offset by concomitant increases in autochthonous detrital production. Reduced riparian forest canopy increases light levels and therefore increases algae production. The abundance and composition of detritivore (macroinvertebrates that process detritus) assemblages in streams are determined in large part by the plant composition of riparian zones (Gregory et al., 1991). Therefore, by changing the stand composition, the macroinvertebrate composition may be altered.

For this analysis, the FEMAT (1993) guidelines of approximately 0.5 SPTH to provide most protection of leaf and litter inputs was used as the criteria to compare alternatives. The full estimated curve displaying the assumed relationship is displayed in Section 3.4 of the EIS.

Microclimate

Literature Review

Microclimate is a collection of variables that are highly dependent on local conditions; hence, microclimates tend to vary greatly across the landscape and over time. Important components of microclimate include solar radiation, soil temperature, soil moisture, air temperature, wind velocity, and air moisture or humidity (Chen, 1991; Chen et al., 1992; Cadenasso et al., 1997). Changes in microclimatic conditions within the riparian zone resulting from removal of adjacent vegetation can influence a variety of ecological processes that may affect the long-term integrity of riparian ecosystems (Spence et al., 1996). For example, microclimate may influence water quality, particularly water temperature (Brosofske et al., 1997), which affects fish. Additionally, microclimate is known to be important for stream/riparian species other than fish, such as amphibians (see Section 3.8).

Characteristics of microclimates in and around riparian lands can be very different from that of upland areas due to differences in vegetative communities, disturbance frequency and intensity, and hydrologic processes (OFPACSW, 2000). In general, due to their low-lying position on the landscape, riparian areas tend to be cooler than the surrounding hillslopes, especially during the night. Because riparian areas are adjacent to water bodies, they often have higher relative humidities under the canopy than similar upslope areas. This increase in humidity combined with shading can cause intact forested riparian areas to have a moderating effect on microclimate (Beschta et al., 1995).

The type and height of vegetation present can influence forested microclimates in both upslope and riparian areas. Below forested canopies, temperature increases with height above ground while relative humidity displays an inverse relationship to this pattern. These patterns will vary depending on species, spacing and tree height (Geiger, 1965 in Beschta et al., 1995). In the winter months, a coniferous canopy maintains a warmer air temperature relative to the surrounding uplands and a cooler air temperature during the summer. In contrast, a deciduous canopy provides a warmer air temperature in the summer relative to a coniferous canopy, but a cooler air temperature relative to an open area. Therefore, it is logical to hypothesize that species conversion from conifer to hardwood would influence the seasonal microclimate of riparian areas (Beschta et al., 1995). Conversely, conversion from hardwood stands to coniferous stands would tend to moderate temperatures along riparian zones in both seasons.

In western Washington, Brosofske et al. (1997) found riparian microclimatic gradients existed for air temperature, soil temperature, surface air temperature and relative humidity. Air and soil temperature varied in a non-uniform way (referred to as gradients) across the riparian zone and into the adjacent uplands. These parameters approached upland forest interior values within 100 to 155 feet from the stream, although surface temperature and humidity gradients often extended further (100 to 200 feet). Harvesting interrupted or eliminated the riparian microclimatic gradients. As a result, the riparian microclimatic gradients approximated clearcut values after harvest, instead of the pre-harvest forest interior gradients. The study concluded that no-harvest buffers of at least 148 feet wide on each side of the stream are needed to maintain natural microclimate environments near streams; but that up to 985 feet for complete protection of all individual variables. Consequently, many standard buffer widths currently in use do not fully protect riparian microclimate. This study determined that riparian microclimate can be influenced by activities that occur in the watershed outside of the buffered area. The authors (Brosofske et al., 1997) suggested selective harvest instead of clearcutting in upland areas near small streams could help increase the effectiveness of riparian buffers.

Evaluation Criteria

To evaluate the environmental effects of the alternatives on microclimate variables within riparian lands, the recommended buffer widths found in the literature were summarized, and the buffer widths and allowed management practices in each alternative were compared to the recommended widths and practices.

The relationship between buffer width and riparian microclimate has received limited study. Ledwith (1996) in the Mad River Ranger District, Six Rivers National Forest, California, examined the relationship among buffer widths, air temperature, and relative humidity. Ledwith (1996) found that air temperature above the streams increased exponentially with decreasing buffer width. The most significant change occurred between 0 to 100 feet. Relative humidity was inversely proportional to air temperature with the most significant drop also occurring between the 0 to 100 feet collection sites. Both parameters had continuous but less dramatic changes between the 100 and 492-foot buffer width collection sites. To avoid significantly altering the microclimate of a riparian zone, Ledwith (1996) recommended leaving buffer strips over 100 feet wide. Buffer strips wider than 100 feet would still affect the microclimate, but at a lower rate of change.

Chen's (1991) data showed the following depth-of-edge effects: 100 to 295 feet penetration depths for solar radiation, soil temperature, and soil moisture; 590 to 787 feet for air temperature; and greater than 787 feet for humidity. These effects, however, are site-specific and vary with edge orientation and weather conditions (Chen, 1991). FEMAT (1993) used the analysis of Chen (1991) to recommended buffer widths of up to three SPTHs, which is roughly 340 feet. However, because this study was conducted on upland forest, it may not be applicable to riparian zones.

More recently, in a study in western Washington, Borsofske et al. (1997) suggested that a 148-foot buffer width would be sufficient to protect microclimate variables in riparian zones with 70 to 80 percent sideslopes in western Washington. However, they also state that this is probably an underestimate and that for some microclimate variables as much as 985 feet may be required. In a later paper, Dong et al. (1998) indicated that greater than 230 feet would be required to maintain natural air temperature gradients. Like air temperature, humidity appears to require buffer widths greater than most microclimate variables to maintain natural conditions (Chen, 1991). However, the conclusions by Chen (1991) are based upon observations from upslope areas rather than within riparian areas.

Brosofske et al. (1997) indicated that there is a gradient for each microclimate variable across the riparian zone. In this analysis, the effects on overall microclimate gradients are evaluated, with additional consideration given to air temperature and humidity. While there are not, as of yet, recommended buffer widths for maintaining microclimate gradients, the results of Borfske et al. (1997) and Dong et al. (1998) provided crude guidelines to evaluate the alternatives. Based on their work, a minimum of 147 feet is considered necessary to maintain most microclimate gradients, while for air temperature, buffer widths greater than 230 feet may be needed.

Using the available information from Chen (1991) for various microclimate variables, (FEMAT 1993) constructed theoretical relationships between percent effectiveness and buffer widths. Pollock and Kennard (1999) suggested the curve for relative humidity should reach 100% at about 240 feet. The curves from FEMAT (1993) and the relative humidity curve from Pollock and Kennard (1999) are depicted in Figure 3.4-4 and are the basis for our qualitative analysis on the effects of the alternatives on microclimate. Note that these curves are based on no-management (i.e., no-cut) forested buffers.

Water Quality

Water Temperature

Literature Review

Stream temperature is influenced by many factors including latitude, altitude, season, time of day, flow, channel width and depth, groundwater flow, stream shading from topography or vegetation, and coastal fog (MacDonald et al., 1991). Temperature plays an integral role in the biological productivity of streams. Aquatic life is the water use that is the most sensitive to water temperature. Salmonids and some amphibians appear to be the most sensitive to water temperatures. Thus, they are used as indicator species regarding water temperature and water quality. Coldwater species such as salmonids are susceptible to harm when stream temperature is greater than 70F (Oregon Department of Environmental Quality [DEQ], 1995) (see Section 3.8). Juvenile salmon and trout are susceptible to harm when the stream temperature is above 73 to 77F. Bull trout require relatively low temperatures during spawning and egg incubation (Oregon DEQ 1995).

Stream water temperature is regulated by heat exchange between the stream water and the aerial and subsurface conditions. Heat energy is transferred to and from streams by direct solar radiation (short wave), long-wave radiation, convective mixing with air, evaporation, conduction with the stream bed, and advective mixing with inflow from groundwater or tributary streams (Beschta et al., 1987;

Sullivan et al., 1990). Direct solar radiation is typically the dominant source of energy input to streams. Long-wave radiation loss is determined primarily by the temperature differential between water and air, with the greater exchange occurring when the difference between the air and water temperatures is greatest (Spence et al., 1996). Convective and evaporative heat transfer are controlled by temperature and vapor pressure gradients at the air-water interface (Beschta et al., 1987). Conductive heat transfer between stream substrate and water generally represents a minor component of a stream heat budget (Spence et al., 1996).

The role of advection depends upon the volume of groundwater or tributary inputs relative to the total stream discharge. In addition, the downstream velocity of the combined flows will also influence the length of the mixing zone from a stream and its inflows. Eventually, the mixed waters will reach a dynamic equilibrium with local environmental conditions. As groundwater flows toward streambeds, water temperatures equilibrate with those in the subsurface soil layers (Beschta et al., 1987). As a result, the temperature of water that enters streams from groundwater flow depends upon the ambient conditions in the soil environment. Seasonal fluctuations are greatest at the soil surface and decrease with depth to the "neutral zone," generally about 16 to 18 meters below the surface where temperatures remain constant throughout the year (Meisner, 1990). In some circumstances, groundwater seeps in streams may substantially reduce stream water temperatures in localized areas.

All of the above processes interact to produce the temperature regimes observed in streams and rivers. These processes also result in a natural warming trend in streams from headwaters to the ocean (Zwieniecki and Newton 1999, Sullivan et al. 1990). In small- to intermediate-size streams of forested regions, incoming solar radiation represents the dominant form of energy input to streams during the summer, with convection, conduction, evaporation, and advection playing relatively minor roles (Brown, 1980; Beschta et al., 1987; Sullivan et al., 1990). Groundwater discharge to streams may be important to small streams where groundwater discharge provides a large percentage of the overall discharge, particularly in the summer months during low flows. Downstream where the stream is larger, the effects of riparian shading and advective mixing diminish and evaporative heat-loss processes increase. Microclimate is discussed in more detail in Section 3.4 (Riparian). Forest practices affect stream temperature through tree harvest in riparian areas, which reduces shade over the stream and results in increased solar radiation on the stream. A recent study by Zwieniecki and Newton (1999) suggest that after the natural downstream warming trend for a stream is accounted for, streams in coastal Oregon equilibrate to local environmental conditions within about 492 feet.

Evaluation Criteria

Changes in water temperature and light regime have both positive and negative consequences for salmonid production and are often difficult to predict. Removal of streamside vegetation allows more solar radiation to reach the stream surface, increasing water temperature and light availability (Brown and Krygier, 1970; Meehan, 1970; Beschta et al., 1987; Bisson et al., 1988a). The interpretation of much of the early research on water temperature changes induced by logging was that these alterations were predominantly harmful to salmonids (Lantz, 1970).

Water temperature increases can be expected to influence embryonic, juvenile, and adult salmonids in small streams (Hicks et al., 1991). It is likely that effects during the time that juveniles are rearing in freshwater are the most significant. Temperature increases can also affect fish survival by increasing the virulence of many diseases, modifying the effects of toxic materials (Lantz, 1970), and lowering the amounts of oxygen available to salmonids.

The upper and lower limits of temperature tolerance in fish can be extended through both adaptation and resistance (Fry, 1947). Brett (1952) reported that more time was needed for acclimation to low temperatures than to high temperatures. He also determined the lethal limits for high and low water temperatures for the young of all species of Pacific salmon using a range of acclimation temperatures. Coho and chinook salmon were the most tolerant of high temperatures although no species could tolerate temperatures exceeding 25.1°C (77.2°F) for exposure times of 1 week.

Adult salmon and trout respond to stream temperatures during their upstream migrations (Bjornn and Reiser, 1991). Delays in upstream migration due to excessively warm natal streams have been observed for sockeye salmon (Major and Mighell, 1966), chinook salmon (Hallock et al., 1970), and steelhead (Monan et al., 1975). Bell (1986) reported that Pacific salmon and steelhead have migrated upstream at temperatures between 3 and 20°C (37.4 to 68°F).

Salmonids have their highest level of metabolic efficiency within the range of 10-14° C (50-57° F), and growth is reduced at higher and lower temperatures (Bjornn and Reiser, 1991). Growth ceases for coho when temperatures exceed 20.3°C (68.5°F) (Reiser and Bjornn, 1979; Brett, 1952). At temperature extremes, fish not only do not have the energy to acquire food, but they also cannot digest it. The capacity for work, including swimming, declines and fish will eventually starve to death if they do not succumb to some other cause first (Beschta et al., 1987).

Water temperature may affect competitive interactions in several ways. Elevated temperatures may increase competition as fish crowd into cooler areas to avoid high temperatures. In cohabiting situations, Reeves et al. (1987) found that steelhead were dominant in cooler temperatures, while redside shiners (a highly competitive non-salmonid fish) were dominant at temperatures above 19° C (66.2°F). Bull trout appear to be one of the more sensitive salmonids to degraded habitat conditions, primarily due to having fairly restrictive requirements. In freshwater, adult bull trout prefer very cool water temperatures for rearing (less than 55°F) and spawning (less than 50°F; Oregon DEQ, 1995).

Streams can be too cold as well as too warm for upstream-migrating salmonids (Bjornn and Reiser, 1991). Cutthroat and rainbow trout have been observed waiting for tributaries to warm in spring before entering them to spawn. Adult steelhead that return from the sea in summer and spend the winter in inland rivers before spawning in spring, overwinter in large rivers downstream from their natal streams because the smaller headwater streams are often ice-choked during winter. It is believed that steelhead overwinter in the larger rivers because survival rates are higher there and the slightly higher temperatures in the rivers enable timely maturation (Reingold, 1968).

Forest canopy removal has resulted in increased winter temperatures in some coastal drainages of the Pacific Northwest (Beschta et al., 1987). Slight post-logging increases in late-winter water temperatures were found in Carnation Creek, a coastal stream on Vancouver Island, British Columbia (Hartman et al., 1987; Holtby, 1988). These temperature increases led to accelerated development of coho salmon embryos in the gravel and earlier emergence of juveniles in the spring.

Earlier emergence resulted in a prolonged growing season for the young salmon but also increased the risk of downstream fry displacement during late-winter freshets. The juveniles that were able to survive to the rearing stage had a higher proportion of 1-year old smolts rather than the normal high proportions of 2-year olds. Using a marine survival model developed by Bilton et al. (1982), marine survival was expected to decline sharply as the fish were smaller than the normal 2-year olds.

One method for measuring forest canopy as it is related to shade is by considering the angular canopy density (ACD). In contrast to canopy closure, which measures canopy density projected to a horizontal surface, ACD is the projection of canopy closure at the angle at which solar energy passes through the canopy to the stream. In areas intensively managed for timber harvest, maintaining vegetation buffers along the stream banks is an effective way to maintain stream temperatures at levels appropriate for fish and other aquatic organisms.

Specific approaches for managing riparian vegetation to protect water temperature in western Washington are provided by Sullivan et al. (1990) and Caldwell et al. (1991) and are summarized in the watershed analysis training manual (WFPB, 1995b). These sources identified a number of important considerations relative to protection of stream temperature, including: (1) nonfish-bearing waters that contribute 20 percent or more of the volume of fish-bearing Type 1, 2, or 3 waters significantly influence water temperature; (2) water temperature reaches equilibrium with local conditions once streams have traveled for approximately 500 feet through a zone of uniform canopy closure; (3) in western Washington, at elevations greater than 3,600 feet, stream temperature is unlikely to exceed temperature standards, even when timber harvest activities occur; (4) target shade requirements vary with water type and elevation; and (5) for Type 1, 2, and 3 waters, total stream shading of 50-75 percent is generally required to maintain streams within water quality standards.

Water temperatures in Type 4 and 5 waters are more sensitive to changes in streamside shading than Type 1, 2, and 3 waters downstream (Timber/Fish/Wildlife Temperature Work Group, 1990). Cumulative downstream effects of increased temperatures in headwater tributaries have not been documented. It would be expected that, assuming similar amounts of ground water inflow into lower streams, the proportion of Type 4 and 5 waters in a watershed may affect overall downstream water temperature sensitivity in the planning unit.

Caldwell et al. (1991) measured stream temperature increases of 2 to 8^{o}C along harvested small streams (Type 4 waters) compared to similar streams with a mature canopy. Small streams tend to be influenced by air temperature, elevation, groundwater inflow, and shade. Along small stream that have been harvested, substantial amounts of understory shade remain, averaging close to 40 percent shade for the study sites. In addition, the shade reduction along a small stream is a short-term phenomenon, existing for about 5 years in stream reaches not affected by catastrophic flooding. In general, the study concluded from the study sites that the understory shading and in some cases the relatively high proportion of groundwater flow, can tend to depress stream temperatures even though air temperatures in the harvested units are typically higher than those in nearby riparian areas. In addition, stream temperature increases to salmonid bearing stream resulting from timber harvest upstream on Type 4 waters appeared to be negligible after a distance of about 150 meters downstream of the Type 4-Type 3 stream confluence. The results reported by Caldwell et al. (1991) have been criticized because the Type 4 streams were generally much smaller than the Type 3 streams into which they flowed and because all downstream reaches did not have old-growth canopy characteristics (ISR 2000). However, the study by Zwieniecki and Newton (1999) on larger streams generally supports the findings by Caldwell et al. (1991).

However, for small streams (<5 ft wide) that are often completely shaded by woody vegetation and hence have no riparian canopy opening in their undisturbed state, a RMZ width of less than 0.75 SPTH is likely to be sufficient to provide full shade. This assumption is supported by Broderson (1973) which reported that a 50-foot buffer provided 85 percent of the maximum shade for small streams (defined as streams with mean annual discharges of less than 5 cfs). As a result, a minimum of 50-foot buffer was used as the minimum criteria to evaluate the risk of shade reduction along small perennial streams. For seasonal streams that do not flow during the summer stream shade should have minimal to no effect on temperature and therefore will not be considered when evaluating shade requirements.

Rashin and Graber (1992) evaluated the effectiveness of BMPs for protecting water temperatures in streams in western Washington. The riparian management zones studied were narrow and included some partial cutting. The riparian management zone buffers were considered ineffective on many of the streams that were studied, particularly low to moderate elevation reaches, reaches effected by flow losses and beaver ponds. Rashin and Graber (1992) also evaluated the methods of Sullivan et al. (1990) and considered their methods to offer major advantages because the methods included parameters such as site elevation and riparian shade. The results of the Rashin and Graber (1992) resulted in modifications to the model used to predict water temperatures in the Forest Practices Rules manual (Doughty et al 1993). If watershed analysis procedures and requirements (WFPB, 1995b) alone are implemented a low to moderate level of protection for water temperature is anticipated.

Buffer strips approximately 100 feet wide are believed to shade the stream to the same extent as old-growth forests which typically have ACDs of 80-90 percent (Beschta et al., 1987). Other studies, summarized in Johnson and Ryba (1992), generally recommend a similar buffer width of approximately 96 feet to protect stream temperature. If the buffer is less than 100 feet, or if the buffer is selectively logged, considerations such as species composition, stand age, and vegetation density become important (Beschta et al., 1987).

The sensitivity of streams to changes in water temperature may vary regionally. Such regional differences in temperature sensitivity are due to a number of factors including elevation (Rashin and Graber, 1992), distance from the watershed divide, and regional air temperature (Sullivan et al., 1990). Because air temperature decreases with increased elevation, streams at higher elevations are expected to be cooler and less influenced by shade levels than downstream areas. Proximity to the coast may also influence geographic variation in stream temperature although relationships are poorly defined. Data in Sullivan et al. (1990) suggest that coastal streams tend to have higher summer temperatures than streams on the west slope of the Cascades. However, since data on streams with equivalent shading, elevation, and flow are limited, this trend should be considered weakly supported.

The number and type of wetlands in a watershed may also influence stream temperature, particularly during low-flow periods, by augmenting stream flow with cool ground water or well-shaded surface water from wetland outlets and subsurface flow.

As discussed above, many factors can influence stream temperature such as shade, air temperature, and groundwater inflow. However, forest practices can reduce canopy cover near streams. The evaluation criterion for stream water temperature is the protection of streamside shade to maintain stream temperature (no increase). As discussed in Section 3.4.3.1, a no-harvest buffer width of 0.75 of a site-potential tree will be used as the criteria to evaluate the effectiveness of riparian management zones to maintain shade, and thus stream water temperature for streams greater than 5 feet in width (Spence et al., 1996). For streams less than 5 feet in width, the evaluation criteria will include the protection of hyporheic zones (areas where groundwater enters stream), seeps, and sensitive sites in combination with maintenance of a 50-foot no harvest RMZ that provides full shade protection of small streams (Broderson, 1973).

Sediment

Literature Review

Two of the most common water quality parameters measured and monitored for sediment are suspended sediment and turbidity. Both are related to sediment delivery and transport in hydrologic systems. Streams that exceed water quality objectives for sediment would have high suspended-sediment delivery rates and/or turbidity.

Suspended Sediment
Suspended sediment is the portion of the sediment load suspended in the water column. The grain size of suspended sediment is usually less than 1 mm in diameter (clays and silts) (Sullivan et al., 1987).

Turbidity
Turbidity refers to the amount of light scattered or absorbed by a fluid and is measured in nephelometric turbidity units (NTUs). In streams it is usually a result of suspended particles of silts and clay, but also organic matter, colored organic compounds, plankton and microorganisms. It is measured in NTUs. Although turbidity in a stream is highly variable and the relationship between turbidity and suspended sediment must be determined for each watershed, turbidity is regarded as the single-most sensitive measure of the effects of land use on streams, mainly because relatively small changes in suspended sediment can cause a large change in turbidity (MacDonald et al., 1991).

Biological Effects of Sediment
Biological effects of increased turbidity may include a decrease in primary productivity of algae and periphyton due to the decrease in light penetration (see Section 3.8). Declines in primary productivity can adversely affect the productivity of higher trophic levels such as macroinvertebrates and fish (Gregory et al., 1987).

Siltation and turbidity reduce the diversity of aquatic insects and other aquatic invertebrates by reducing interstices in the substrate. Several studies (Nuttall and Bielby, 1973; Bjornn et al., 1974; Cederholm et al., 1978) have demonstrated that species density and diversity drop with increased fine sediment deposition in gravels.

Siltation and turbidity have also been shown to affect fish adversely at every stage in their life cycle (Iwamoto et al., 1978); spawning and incubation habitats are most directly affected (Spence et al., 1996). Deposited sediments tend to have a greater impact on fish than suspended sediment.

Particulate materials physically abrade and mechanically disrupt respiratory structures (e.g., gills in fish) (Rand and Petrocelli, 1985). Sediment covers and fill intergravel crevices, which fish use for shelter, decrease the carrying capacity of stream habitats for young salmonids (Bjornn et al., 1974). Turbidity reduces light penetration, which affects the food capture reactive distance of juvenile and adult salmonids (Spence et al., 1996). Jensen and Newcombe (1996) conducted a metadata analysis of suspended sediment effects to salmon and suggested that both sediment levels and duration should be considered when discerning the level of effect.

Evaluation Criteria

An increase in suspended sediment has numerous effects on the aquatic system. Physically, fine sediment can impair municipal and agricultural use of water, affect bed material size, and alter the quantity and quality of habitat for fish and benthic invertebrates. Fine suspended sediments can also affect the chemistry of the water as chemical nutrients and other chemicals are adsorbed onto fine particles.

Timber harvest activities such as road building and timber yarding may increase sediment input into streams. The key factors controlling sediment increases are (1) the intensity of the disturbance, (2) the areal extent of the disturbance, (3) the proximity of disturbance to the channel system, (4) storm events experienced when the site is most sensitive to erosion and mass movements, and (5) BMPs used to control sediment delivery to streams (Everest et al., 1987; Swanson et al., 1987). Sediment can be eroded from road surfaces, road fills, or slope failures associated with road construction (e.g., blocked culverts). Increased sediment yields tend to be persistent from slope failures and road surface runoff. Timber harvest often results in surface erosion from landings, skid trails, and other compacted areas (Binkley and Brown, 1993; MacDonald et al., 1991; Moring, 1982). Ziemer et al. (1996) note a 400 percent increase in suspended sediment following road building, and a 100 to 500 percent increase after logging commenced for timber harvest in the early 1970s in the Caspar Creek watershed near Fort Bragg, California. They noted much smaller effects for logging that occurred from 1985 to 1991 because of improvements in BMPs (Ziemer et al., 1996). Section 3.2, Sediment Sources, discusses these effects in detail.

Fire can also be a source of increased sediment yield, primarily through increased surface erosion, which is caused by a decrease in protective vegetation and an increase in surface runoff (see Section 3.6).

Timber harvest effects on turbidity closely correspond to the effects on suspended sediment (Barber, 1997; Brown and Ritter, 1971). The same dominant processes that increase suspended sediment will increase turbidity: landslides, surface erosion, and road erosion (see Section 3.6).

Sedimentation includes the processes of erosion, sediment transport, and deposition. Deposition is the temporary or permanent stoppage of sediment movement. Surface water quality is not affected if sediment is deposited before reaching a water body. Once sediment reaches streams, deposition can occur several times over. As flow velocities and volumes increase, sediment is moved downstream. If flow volume or velocities decrease, deposition can occur. The amount of sediment suspended or moved along the streambed therefore depends on surface water movement.

Sediment affects water quality in several ways. It creates a turbid (muddy) condition that restricts light in the stream environment. Nutrients combined with, or attached to, the sediment particles are added to surface water. Oxygen-demanding materials associated with sediment can reduce dissolved oxygen content. Sedimentation may also introduce harmful minerals into surface water.

The evaluation criterion for sediment-related water quality parameters is the overall reduction in sediment delivery to streams from management activities. These include reduction in chronic erosion sources such as surface erosion and episodic sediment such as landslides from BMPs for timber harvest, road construction, road use, road maintenance, and road abandonment. Table 3.2.1 summarizes the reduction in sediment delivery to streams by source and alternative.

Pesticides

Literature Review

Pesticides used in forest management include a wide variety of chemicals introduced to the forest environment with the intent of controlling or halting the proliferation of nuisance organisms. Pesticides are commonly grouped according to one of three target organisms: plants (herbicides), insects (insecticides), and fungi (fungicides). The active ingredients usually determine the effects of individual pesticides. In addition, prior to application, almost all pesticides are combined with a surfactant (i.e., a surface-active agent) or other adjuvant (i.e., a pharmacological agent added to increase or aid the pesticide's effect) to control and improve the desired effect. Although these additives present lesser threats to the environment than the active ingredients in the pesticides, their impacts can be significant.

Once released into the forest environment, pesticides can have a wide range of fates and impacts, depending on their specific chemical properties, the methods and conditions of application, and the environmental conditions into which they are introduced. The purpose of the Forest Practice Rules on pesticide applications is to "regulate the handling, storage and application of chemicals in such a way that the public health, lands, fish, wildlife, aquatic habitat, and water quality will not be endangered by contamination" (WAC 222-38-010). However, this report primarily focuses on impacts to water resources as the alternatives are nearly identical with respect to impacts to lands and terrestrial species.

Pesticides used in the forest environment can become water contaminants if they are transported to surface waters or groundwater. Transportation to surface waters would most likely occur through wind drift; however, heavy rains can result in pesticide transport in surface runoff or erosion. Pesticides can also be directly applied to surface waters by overspray and spills. Groundwater contamination can occur through contaminated surface water recharge and through the direct transport of pesticides from the soil surface by rainwater.

Data are available regarding pesticide levels found in Washington State streams and groundwater. However, only a few studies have focused directly on forestry applications. Most studies have instead focused on pesticides used for agricultural production. Likewise, because there is some overlap between the two practices, it is not possible to determine which pesticides originated from agricultural sources versus those contributed from forested land. Nonetheless, pesticides that have been detected in streams and groundwater are present at very low concentrations, usually well below applicable water quality standards. However, some pesticides have been detected at concentrations that exceed the more restrictive guidelines for the protection of aquatic life (freshwater-chronic criteria) or health advisories for drinking water. Although few studies have focused directly on forest chemical applications, the available data and research do indicate the potential for certain chemicals to enter and persist in surface waters and groundwater supplies.

For more details on the types of pesticides used on forested lands and the extent of the use of forest pesticide applications in Washington State, see Appendix J. This appendix also includes information on the specific pesticides used, their toxicity, mobility, persistence, and carcinogenocity.

Evaluation Criteria

Pesticides have the potential to contaminate surface waters and groundwater depending on the amount of pesticides applied, the application technique, and the environmental conditions under which they are applied (such as ambient wind speed or soil moisture content). The evaluation criteria for pesticide applications focus on how well the Forest Practices Rules protect water resources from contamination resulting from pesticide applications (e.g., spray drift, runoff, erosion, seepage to groundwater, etc.). In addition, the evaluation criteria take into account how well the alternatives protect riparian plants from damage caused by pesticide applications. Finally, the criteria also consider the potential impacts to fish and aquatic wildlife resulting from contamination of water resources.