46

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Assessment of Cumulative Effects on Salmonid Habitat: Some Suggested Parameters and Target Conditions

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INTRODUCTION

Cumulative effects (CEs) as used in this paper refer to changes in watershed and channel conditions caused by multiple forest practices. These effects may be additive or multiplicative in nature and are functionally linked to watershed processes.

Public concern and legal redress of CEs due to forest management in Washington date back to the late 1970's. Golde et al. (1989), provide an excellent historical review of how the Washington State Forest Practices Board has dealt with CEs evaluation and regulation on state and private forest lands. Efforts to address CEs through the 1986 Timber Fish and Wildlife Agreement (TFW), stressed three approaches; development of basin plans, setting resource goals and monitoring management practices in non-plan basins, and by encouraging multi-disciplinary cooperative research. Basin plans became known as Resource Management Plans (RMP), and among other things these plans were to set resource goals at any appropriate geographic scale and monitor to determine if those goals were being met. The approach was to emphasize adaptive management in the evaluation of resource goals and risk assessment.

Two early RMPs, the Upper Yakima and the Nisqually plans, established resource goals for in-stream habitat parameters or "thresholds", as they became known, to indicate relative habitat condition for fish. If a particular parameter was found to be out of the agreed upon bounds, prescriptions were to be applied to forest practices that in theory had linkages to channel inputs and processes responsible for it's present state.

The concept of linking threshold values for selected in-stream habitat parameters to the intent and degree of forest practice regulation, has gained favor among many resource managers and is now being considered for adoption into the Forest Practice Rules and Regulations. Thresholds have been variously called decision criteria, performance standards, habitat goals, target values, and most recently "indices of resource condition". Whatever their name, their purpose is the same; to describe a level of stream habitat condition that would trigger specific management responses. A description of these responses is still under development in the draft rules for "Cumulative Effects and Watershed Analysis", Washington Administrative Code (222-31-030 to 222-32-100). We prefer the term "target condition" because it connotes a condition that may be surrounded by considerable variability but that can be actively managed for.

The objectives of this paper are: 1) to provide an evaluation of the "threshold" approach as it might apply to the regulation of forest practices across diverse landscapes and stream channels in Washington, 2) identify in-stream parameters that are closely linked to forest practices, fluvial processes and salmonid habitat life history requirements, 3) suggest threshold or target conditions to determine habitat condition, and 4) recommend quantitative field methodologies for parameter measurement.

The organization of this paper is intended to serve readers of various interest levels and disciplines. Important conclusions and recommendations are presented first, followed by an increasingly detailed discussion of supporting information.

CONCLUSIONS AND RECOMMENDATIONS

1) Reliance on in-channel parameters and resource based target conditions will not prevent CEs from occurring. In effect, this approach permits CEs to occur before corrective actions are undertaken. Such a system is helpful only in assessing channel condition and it philosophically continues to place the burden of proof on fishes and other aquatic resources to show damage. This is not appropriate in view of the lag time between some watershed activities and their expression in the channel, and it is not entirely consistent with the goals of Federal legislation such as the Clean Water Act and the Endangered Species Act.

2) A better approach to the prevention of CEs is an early warning system of hillslope thresholds that would deal directly with the causes of CEs on forest lands, i.e. those things that upset the balance of basic channel inputs of wood, water and sediment. These include clear cut logging on steep unstable terrain; road density, location, construction and maintenance; and patterns and rates of timber harvest that disrupt normal basin hydrology.

3) Approaches to the prevention of CEs, that do not deal with the proximal causes of stream habitat deterioration as the first level of warning, fail to provide adequate resource protection and do not serve forest managers because they provide a false sense of security and ultimately limit management flexibility to deal with the problem.

4) A suite of in-channel parameters and target conditions is a needed channel assessment procedure that could help identify specific needs to address through forest practice regulation. This system would be a useful adjunct to a hillslope threshold approach and could provide needed corroborative links for monitoring the effectiveness of management practices over long periods.

5) We have elected to use the conditions indicative of the streams draining unmanaged forests as the standard by which to set target conditions. This approach does not seek to optimize the stream environment for a particular species or age class, but assumes naturally functioning and ecologically intact channels will provide long term sustainability for diverse fish assemblages.

6) We have elected to set a single target condition for two reasons, first we do not think it is necessary to designate a lower level for the purposes of channel assessment and secondly the data does not exist to do this in a consistent fashion.

7) The scientific literature does not support a "one size fits all" approach to the establishment of target conditions. We believe a stream classification system linked to important physical variables such as gradient, stream size, and bank material must be developed to group streams and stream reaches that may respond similarly to disturbance. We support the establishment of target conditions appropriate to local settings for assessment purposes.

8) We recommend the following in-channel parameters for inclusion in a detailed monitoring program; a) LWD frequency (pieces per channel width), b) average LWD volume (cubic meters per piece), c) percent pool area, and d) substrate composition (% fines < 0.85 mm).

9) Target conditions established at this time are:

A) LWD frequency by channel width. From 4 to 19 meters bank full channel width, 2.44 to 2.03 pieces respectively (see Table 2a, column 1). Data based on regression of Bilby and Ward (1989).

B) LWD average volume per piece by channel width. From 4 to 19 meters bank full channel width, 0.25 to 3.70 cubic meters per piece respectively (see Table 6, column 1). Data based on regression of Bilby and Ward (1989).

C) Percent pool area. For streams less than 3% gradient, the target condition is for 50% of the total wetted surface area to be comprised of pools at the low flow period.

D) Substrate composition. The target condition is for no more than 11% of the particle size distribution to be comprised of the <0.85 mm fraction. This target condition applies broadly to streams of different sizes and gradients but as a general rule would be most applicable to streams <3% gradient and between 5 and 30 meters bank full channel width.

GENERAL DISCUSSION

We present here a brief discussion of watershed and stream channel responses to logging related disturbance and how fish respond to these habitat alterations. The purpose of this discussion is to develop a thread of continuity for the selection of parameters and target values. In no way is this discussion intended as a full treatment of the subjects. Comprehensive review articles can be found on these and related topics in Meehan (1991).

Watershed and Channel Response to Disturbance

Viewed simply, two things occur during logging; roads are constructed and vegetation is removed. These activities, multiplied across a watershed landscape alter the character or quantity of the basic inputs of wood, water and sediment to stream channels. Changes in the character, timing and quantity of these basic channel inputs initiate adjustments in channel morphology that may persist for decades (Madej 1978). Channels reach new balances between sediment and flow regimes by adjusting a number of key parameters including; width, depth, meander waver length, slope and sinuosity (Schumm 1971). Without additional information about the watershed, however, it is difficult to predict which parameters a channel may adjust in response to a given change of inputs or to predict which inputs have changed based on observed channel adjustments (Nunnally 1985).

Episodic large inputs of wood and sediment to channels, delivered through landslides and channel failures, have been a part of natural channel development in the Pacific Northwest and may even be important in forming productive fish habitat. However, numerous studies have demonstrated that clear-cut logging in steep terrain has often increased the frequency and distribution of these events, probably well beyond the point where they may be beneficial (Fiksdal 1974; Leopold 1980; Schlichte et al. 1991). The recovery time between natural disturbances caused by cataclysmic storms or fire, on the order of centuries, was considerably longer than that observed in many of today's clear-cut logged watersheds.

The overall effect of less large woody debris (Grette 1985, Bilby and Ward 1991), more sediment (Everest et al. 1987) and more frequent channel forming flows (Chamberlain et al. 1991), has been simplified stream habitat (Hicks et al. 1991). Within-basin variability of response should be expected based on local conditions along the length of the channel network. The particular channel setting; including valley gradient, degree of channel confinement, basin size, climate and geology determine the general potential character of habitat while the cycle and magnitude of land-use triggered disturbance determines the proximate habitat response.

Fish Response to Habitat Alteration

The response of salmonids to habitat alteration has been extensively studied. While the scientific literature is full of numerous examples of how fish respond to single habitat conditions, such as temperature (Martin et al. 1986), increased food supply (Mason 1976), habitat structure of various sorts (Murphy et al. 1985, Shirvell 1990); attempts to integrate the contending effects of different habitat variables on salmonid stock size (Holtby and Scrivener 1990; Shirvell 1989), community structure (Fausch et al. 1990), and species interactions (Reeves et al. 1987) are few. Here, we borrow from Hicks et al. (1991) by ordering our discussion first to effects of single habitat variables and then the more complicated integration of simultaneously varying conditions.

Temperature and light

Increases in light and temperature can have both negative and positive effects on salmonids. These two variables usually co-vary due to canopy opening and modify the trophic structure of the stream community, indirectly affecting growth and survival of fish. More light and warmer water stimulate algal growth and shift the primary production away from the diatom dominated community of closed canopy streams. In response to this new food source secondary production shifts to invertebrate species more likely to enter the drift and become food for fish (Hawkins et al. 1982; Gregory et al. 1987).

If summer stream temperatures exceed the range of efficient metabolism, growth rates are reduced. Slightly warmer water temperatures in the winter, however, which is a typical response of low elevation coastal streams to logging (Beschta et al. 1987), can have a positive effect on winter growth and presumably survival to seaward migration. These same winter temperature increases can accelerate intra-gravel development causing fry to emerge earlier from the gravel than they normally would (Holtby and Scrivener 1989). The combination of a longer growing season and better growing conditions can indirectly affect the timing of important life history events and alter the survival through critical life history periods.

Large woody debris (LWD)

Large woody debris plays a vital role in maintaining the distribution and frequency of many diverse flow and cover conditions in small forested streams and in serving to ameliorate the erosive forces of channel forming and flood flows. It is the condition created by the LWD e.g. variable velocity regimes, darkness, and overhead shelter, that fish seek out, and not the structure itself (Shirvell 1990). Juvenile coho salmon and older age classes of steelhead and cutthroat trout strongly prefer the low velocity habitats various kinds of debris-formed pools provide (Bisson et al. 1982). For these salmonids a loss of pools means almost a proportional decrease in their abundance. Seasonally, velocity shadows cast by woody debris may be even more significant in maintaining salmonid abundance (McMahon and Hartman 1989).

Sediment

Researchers have used a dizzying array of methods, analyses and textural indices to investigate the effects of fine sediment on the reproductive success of salmonids (Chapman 1988; Kondolf 1988). There seems little doubt that high proportions of fine sediment in spawning gravels impairs intra-gravel survival of salmonid embryos. Precisely what those proportions are, and perhaps more significantly what the dangerous grain sizes are for different species is still somewhat in question. Chapman (1988) has provided a comprehensive review of the information and concluded that there is difficulty in applying many of the studies to natural stream settings because; 1) laboratory studies failed to duplicate the architecture of natural egg pockets and, 2) field studies had not related survival to emergence to actual egg pocket conditions.

The effects of coarse sediment deposition on fish are more indirect, operating through sometimes subtle shifts in channel morphology that may persist for decades (Madej 1978). The general response of alluvial channels to widen and shallow with higher sediment loads, causes loss of pool volume and in extreme cases, surface flows where channels are actively aggrading. These adjustments directly reduce the rearing space available of salmonids during the summer growing season.

Flow

The effects of altered stream flow are difficult to isolate from other effects of forest management. Although immediately after logging there may be a temporary increase in base summer flows and an increase of summer rearing space, these positive effects may be quickly overshadowed by the negative effects of larger magnitude winter flows. Indirect habitat changes such as the redistribution of LWD and alterations to the channel geometry may be long lasting effects. Immediate effects are increased frequency and depth of streambed scour, attendant loss of incubating eggs (Poulin and Tripp 1986a), and physical displacement of juveniles. These conditions may be exacerbated in the transient snow zone (Harr 1986) where specific storm conditions may combine to produce rapid and large runoff. In rain dominated regions, the primary effect seems to be an increase in the frequency of channel forming flows which cause channel adjustments and form more hostile environments for stream biota.

Combined effects

In the most comprehensive study to date of the effects of logging on salmonid populations, effects of changed habitat conditions, fishing mortality and climate were studied through a series of linked regression models (Holtby and Scrivener 1989). Results for coho salmon indicate slight intragravel water temperature increases during the incubation period accelerated intragravel development causing earlier fry emergence and giving the fish a longer growing season their first summer (Hartman and Scrivener 1990). As a result, average fish size going into the winter was larger, resulting in better over winter survival rates, which produced more 1+ smolts and fewer 2+ smolts (Holtby and Scrivener 1989). Offsetting positive and negative effects made it impossible to measure a direct effect by just looking at stock size at the adult stage. These cascading effects on size and timing of important life history events and population age structure were a surprising result and are a prime example of the complexity of stream ecosystems.

APPLICATION OF PARAMETERS AND TARGET CONDITIONS TO THE REGULATORY ARENA

Damage Assessment versus Prevention

A suite of in-stream parameters and target conditions is a valuable tool for assessment, but by itself, can only be used for evaluation and is not an early warning system for the prevention or remediation of CEs. As such it will serve more as a regulatory lever than a preventative screen. While we believe it is vitally important to understand the condition of the channel network in forested watersheds, the emphasis must always be outside the channel in developing regulatory alternatives to prevent or remediate CEs. Regardless of the condition of the channel, if there are risks on the hill slope, they should be assessed and realistically dealt with, not exacerbated by management activities.

The ultimate value of an in-channel assessment program may be to encourage the development of regional databases that link channel condition to remotely detectable landscape alterations and natural watershed sensitivities, e.g. road density and indicators of hydrologic maturity such as stand age distribution, and unstable soils and slopes. If these general relationships can be developed, hill slope parameters could be substituted for the in-stream parameters in a true "threshold" fashion, to provide clear and early indicators of CEs.

Use of Parameters and Target Conditions

Even if the condition of the stream channel is coupled with a hill slope risk assessment, (as envisioned in the draft forest practice rules at the time of writing), to direct the intent or degree of forest practice prescriptions, those prescriptions must be entirely successful to avoid gradually pushing the habitat conditions from good, to fair, to poor. In the present draft rules, channels judged to be in good condition would, in all but a high risk scenario, be relegated to standard forest practice rules. This approach does not take into account the loss of buffering capacity against the effects of large natural storm events that the standard rules may permit, nor does it recognize the lag time between landscape alteration and it's expression in the channel.

If an evaluation of channel conditions must show an arbitrarily set level of deterioration before management responds to hill slope risks on a priority basis, it is a formula for gradual habitat loss and legitimizes placing the burden of proof on fishes and other aquatic resources to prove damage. This is not entirely consistent with federal legislation such as the Clean Water Act and the Endangered Species Act. Regardless of channel condition, management responses should deal responsibly with risks to public resources.

Dealing with Spatial and Temporal Variability

Regulatory systems have little patience with complexity. However, the scientific reality is that no two streams are alike and they represent a continuum of physical form, energy inputs and biological processes (Vannote et al. 1980). We can however, identify patterns of form and process that will enable us to adequately group streams for management purposes. This is an urgent need and addresses our understanding of stream systems, their potential to produce different fishes and our ability to manage that potential in the face of large scale and perpetual landscape alteration. Recent work by Bradley and Whiting (1991), and other work in progress, (Dave Montgomery pers. comm.) should be invaluable in providing a classification system that helps partition out the inherent variability of different physical settings from that imposed by management practices.

Most of the recent attempts to classify streams have used hierarchical approaches that build linkages between regional and microhabitat scales (Naiman et al. 1992). The TFW Ambient Monitoring Program has had considerable experience with such a system developed by Cupp (1989) for forested streams in Washington. Analysis of stream survey data indicates that the current classification system may be ill suited to adequately account for differences in habitat due to inherent properties of the setting (Ralph et al. 1991). These problems seem to arise from the lack of a truly systematic categorization of channel types based on a continuum of channel characteristics important to fluvial processes. Suggestions for increasing the rigor of a landscape or watershed classification scheme have been made by Orsborn (1990) and Ralph et al. (1991).

Successful implementation of a parameters and target condition approach will require appropriate stratification of streams to ensure realistic expectations for target conditions. The other alternative is a "one size fits all" approach, which could easily confuse inherent variability with management effect. We identified the following characteristics as strong determinants of important in-channel parameters: 1) gradient, 2) some measure of stream size (basin area or channel width), 3) degree of confinement and bank material, 4) basin geology, and 5) regional climate. Not all of these variables are equally important to all parameters and we envision a stratification scheme unique to each parameter for which target conditions are set.

PARAMETER SELECTION

We were asked to evaluate temperature, large woody debris, gravel composition, gravel stability, "primary pool" frequency and cobble embeddedness as parameters for evaluating CEs. Their consideration has been a priority, leaving little time for the examination of other parameters. However, we do not believe there are other parameters for which target conditions could be set at this time. In order to detect CEs across a broad range of stream settings, it will be necessary to use a suite of parameters. For example it would be misleading to use gravel composition as an indicator in a site where the bed has become armored as a result of the loss of woody debris or increased high flows.

MacDonald et. al. (1991) states, "An ideal parameter for monitoring the impacts of a land management activity such as forestry should: 1) be highly sensitive (responsive) to the management action(s), 2) have low spatial and temporal variability, 3) be accurate, precise, and easy to measure, and 4) be directly related to the designated uses of the water body." As MacDonald et. al. note, these idealized parameters do not exist.

SETTING TARGET CONDITIONS

The main assumptions of a parameters and target conditions approach can be summarized in this way: "The most productive conditions for our streams have been determined, they can be represented by a few simple surrogate variables, measured with reliability and watershed activities managed precisely enough to predictably affect the direction and intensity of those variables". Full agreement with this statement requires an uncommon degree of confidence. On the other hand, reluctance to apply what we know, adaptively, would limit our ability to manage our natural resources progressively.

Setting target conditions for specific habitat parameters requires first that an approach be established and secondly that numbers be derived from that approach. Two general approaches are available to managers. The first sets target conditions based on some known or assumed habitat condition that is predicted to maintain a given level of resource abundance. The second approach, and the one we favor, sets target conditions based on streams draining unmanaged forests. A discussion of both follows.

Fish Response

Setting target conditions based on fish response assumes that target levels for specific habitat parameters above or below which aquatic resources would be compromised are known or can be determined with certainty for a wide range of stream settings. We do not believe this to be entirely true. Even if the basic shape of the response functions for different habitat parameters, could be accurately described it still begs the question of whether we can manage precisely enough to maintain any correspondence with the desired levels.

Hypothetical response curves are depicted in Figure 1, each suggesting a different level of risk associated with incremental habitat change. Curve A suggests that a target condition could be set that would allow considerable variation in habitat condition without sustaining a resource loss, B suggests that even small incremental changes in habitat condition could cause significant loss of resource, and C simply requires that arbitrary target conditions be selected and accepts a certain level of impact. Even though these curves are meant to represent generic responses, they could reasonably represent generalized responses to a decrease in pool volume, an increase of fine sediment in spawning gravels and LWD loading for A, B, and C respectively. Each response requires a different level of management precision to maintain a desired level of resource productivity. Since many of the important in-stream parameters co-vary and are not independent, it would be highly unlikely that they could be managed independently. A major problem with this approach is that there is often significant lag time between landscape alteration and the manifestation of it in the channel.

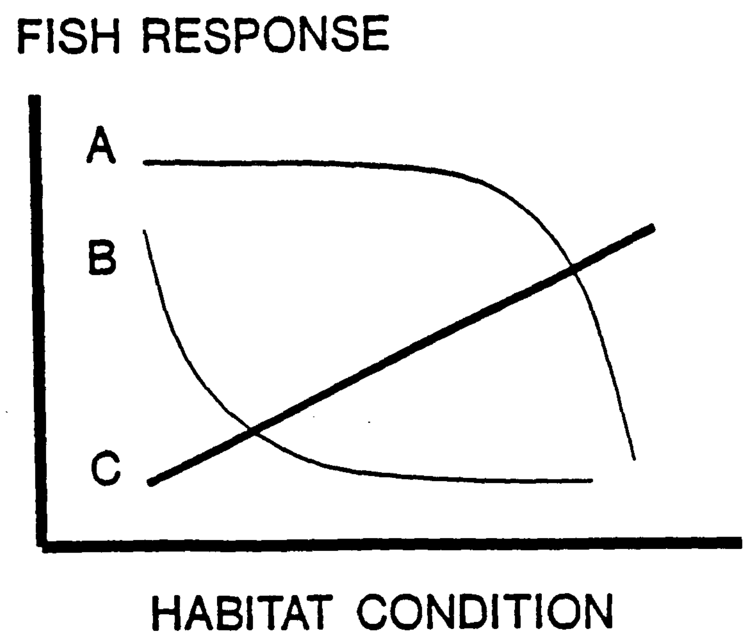


Figure 1: Hypothetical response of fishes to changes in different habitat parameters.

This approach continues to place the focus of our regulatory program on the resource rather than on the processes that support that resource. As a result the resource continues to shoulder the burden of proof to show damage narrowly focusing on specific life history stages of individual species while ignoring the need to provide diverse habitats for whole assemblages of fishes.

Habitat Suitability Indexes (HSI)

HSI methodology is a second way that has been proposed to set target conditions based on fish response. HSI is another way of describing the kinds of habitat conditions that fish favor and thereby predicting some level of use for stream reaches exhibiting those characteristics. The generalized Suitability Index (SI) curves are derived from various studies that demonstrate preferences of individual species and life history stages for particular conditions. The curves are normalized so that each habitat condition can be described on the basis of a standard preference level.

Without calibration to local conditions these curves are relatively inept at describing the extent of habitat utilization for individual species. Even after calibration they are often poor predictors of standing stocks (Wesche et al. 1987). In comparing six mathematical models of the relationships between fish populations and their habitat, (some of which were similar in construction to the HSI techniques), Shirvell (1989) found that for data sets from which they were not derived, they only explained 7-30% of the variation in fish numbers or biomass (mean = 24%). He concluded that, "...they will not prevent management decisions which result in undesirable ecological consequences unless their appropriateness is confirmed before each application."

A principle drawback of HSI methods for application to the regulatory arena is that they do not discriminate between natural setting variability and that attributable to management activities. A suite of variables applied broadly to streams of different gradients, sizes and basin characteristics may give fairly different suitability ratings. This "one size fits all" approach yields conflicting views of true habitat condition. The principle appeal of the method is that it offers a convenient way to select an arbitrary level of suitability, (1.0 being the most suitable and 0.0 the least). Because of the normalized SI curves, these ratings transfer broadly to all HSI variables making it is easy to pick multiple suitability levels to describe varying degrees of habitat condition. However, these selections are purely arbitrary and may have little bearing on actual habitat condition or utilization by fish.

Figures 2 - 5 illustrate the difficulty in applying SI curves even for a single species and age class within the same stream. The overall juvenile coho preferences for velocity and depth as depicted in Figures 2 and 4, (a value of 0 on the Y axis indicating selection for conditions in the same proportion as they occur in the stream), shows preferences for velocities < 10 cm/s and complete avoidances of velocities > 30 cm/s, and preferences for depths > 10 cm deep. However, if behavioral characteristics of the population are considered, those fish that hold territories, "defenders" and those that "wander" have different velocity and depth preferences (Figures 3 and 5). Unless specific information about population characteristics and precise distributions of HSI variables are known for a variety of habitat types within the same stream, application of generalized SI curves can be misleading (Bisson and Fransen unpubl. data).

<u>Ecosystem</u>

A fundamental question about target conditions is; "What do we want streams in managed forests to be like?" We believe the answer to this question is that they should approximate those streams draining unmanaged forests. Because these are the conditions that sustained ecologically diverse communities and healthy salmonid populations over long periods of time prior to cultural development, they are the most secure and represent a unifying basis to evaluate channel conditions. At this time to set target conditions by any other standard assumes, that another standard is; a) more desirable for public resources, or b) represents an acceptable level of impact.

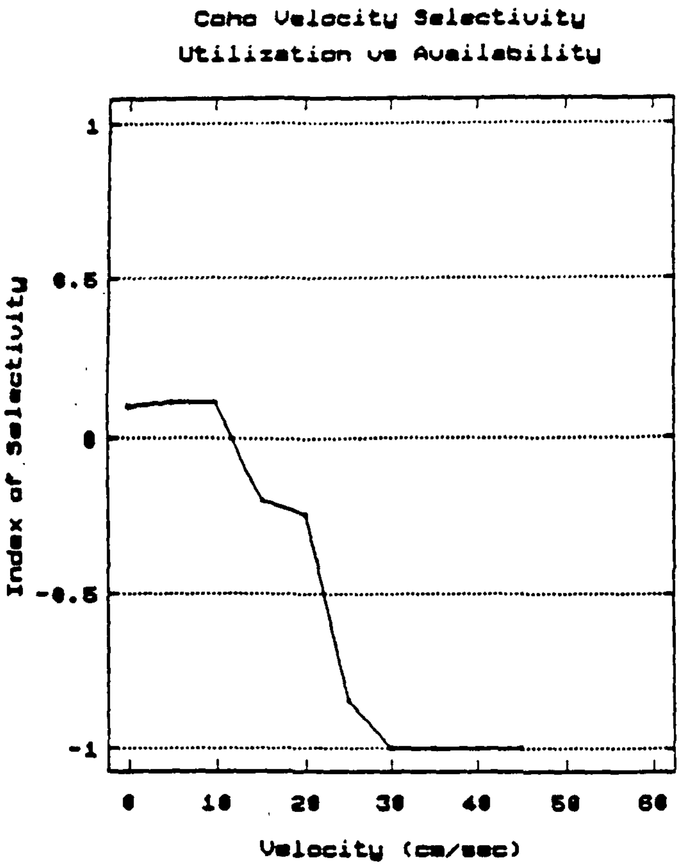


Figure 2:

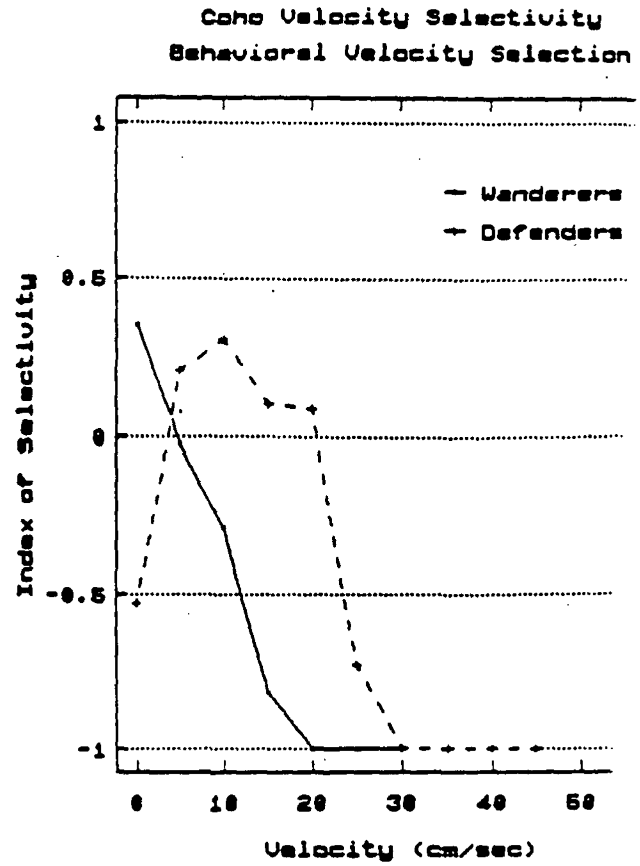


Figure 3:

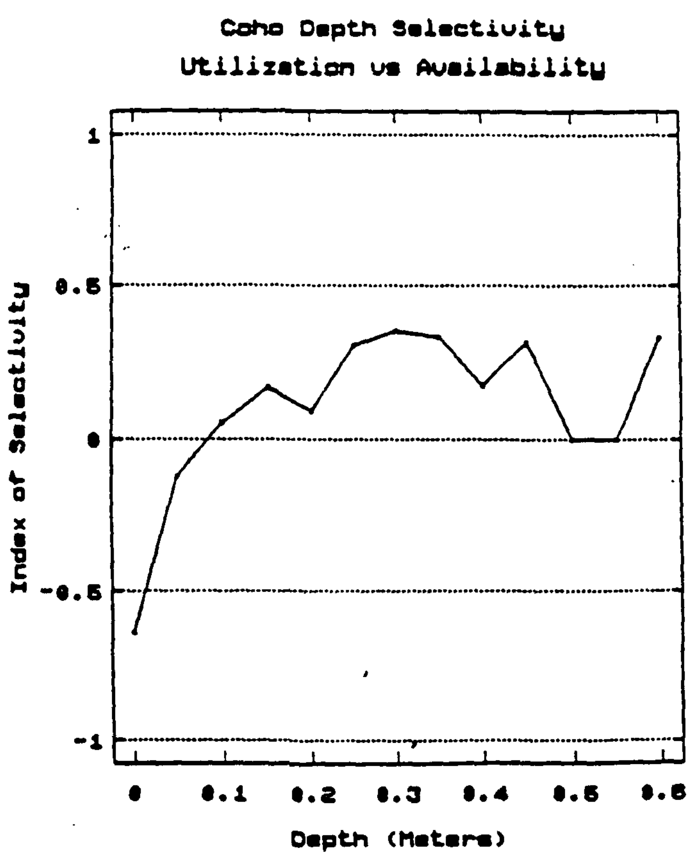


Figure 4:

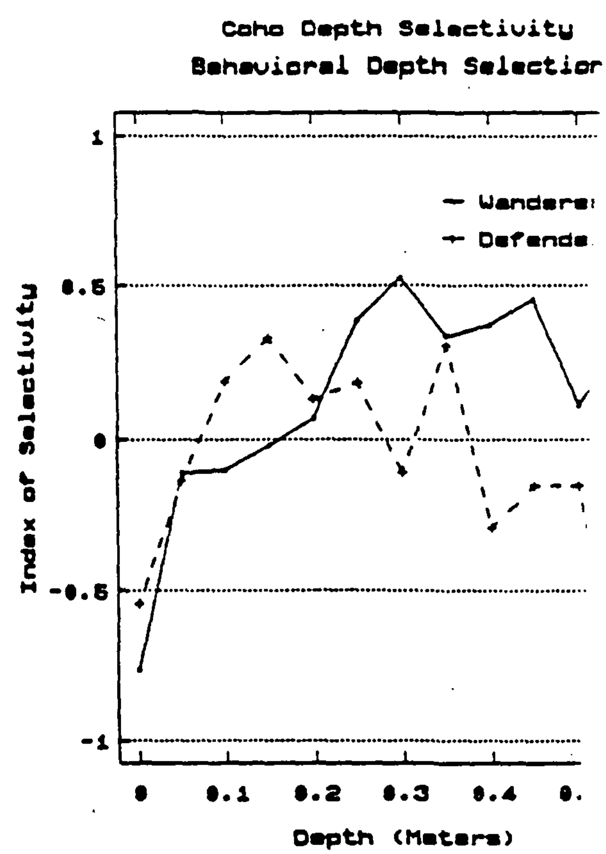


Figure 5:

We believe setting target conditions in this manner, appropriately shifts the focus away from the numbers and arguments that continually place the burden of proof to show damage on fishes and other aquatic resources. The ecosystem approach assumes that suitable conditions for all species and life history stages will be supplied by providing them the stream conditions under which they evolved. Diversity and variability are the norm in unmanaged basins. We should be seeking similar patterns in managed basins. This approach does not seek to optimize the stream environment for particular species or age class but assumes that the conditions in streams draining unmanaged forests will support diverse fish assemblages. These conditions may even be less favorable for some selected salmonid species than those in managed systems. We are willing to assume those risks in favor of long term stability.

The ecosystem approach seeks to provide a balanced basin hydrology, and sediment regime and provide a loading of woody debris similar in frequency, volume and species composition to unmanaged basins. This approach assumes the strong spatial variability of habitat parameters found in unmanaged settings is desirable for long term maintenance of diverse habitats. This is not necessarily a hands off approach to management. To do nothing but wait is probably the longest route to recovery in many cases. This approach can shift the focus of debate on forest management from resource to process, and from protection to recovery.

Determining Multiple Levels for Target Conditions

Describing more than a "good" target condition for managed streams, (which we are designating the mean or better), for unmanaged streams of similar characteristics, (eg. similar gradients, size and bank material), requires arbitrary decisions. Several approaches have been suggested, including the arbitrary selection of an HSI suitability index such as .7 or .8, picking a level where "fish survival is significantly affected", or from the ecosystem approach, some deviation from the mean.

The only pressure to set this lower limit, a level that would designate the difference between "fair" and "poor" conditions, comes from a particular logic that is being institutionalized in the proposed forest practices rules. This logic holds that if a channel is in a certain condition, there are corresponding levels of risk that are permissible on the hill slope. We would rather encourage a logic that independently deals with hill slope hazards, seeks to understand proximate causes for any deviation in channel conditions from the mean values for unmanaged streams, and creatively manages for recovery of the watershed processes that support ecologically diverse channels.

We have elected not to set "poor" target conditions for two reasons, first we do not think it is necessary and secondly the data does not exist to do it in a consistent fashion. Target conditions are standards against which to compare. We believe this approach was effectively applied by Beechie and Wyman (1992).

SUPPORTING RATIONALE FOR PARAMETERS AND TARGET CONDITIONS

Temperature

Target values have been established through state water quality standards and the TFW temperature model (Cumulative Effects Thresholds RFP).

Large Woody Debris (LWD)

The amount of large woody debris (LWD) (also termed large organic debris and coarse woody debris) in streams has been related to salmonid abundance and distribution (Murphy et al. 1985b; House and Boehne 1986; Shirvell 1990) and this relationship has often been attributed to the use of LWD as cover (Bustard and Narver 1975a; Grette 1985; Heifetz et al. 1986; McMahon and Hartman 1989). However, Bjornn et al. (1991) have indicated that the abundance of age-0 coho during the summer in several streams of southeast Alaska was not correlated to cover. These authors however, do not refute the benefit of cover to salmonids during winter. Recent research suggests that salmonids associate closely with LWD primarily to take advantage of the depth and velocity profiles these accumulations create in the channel (Tschaplinski and Hartman 1983; Shirvell 1990; Bjornn et al. 1991; Bozek and Rahel 1991). The role of LWD in the maintenance of pool depth, pool area and hydraulic complexity (Keller and Swanson 1979; House and Boehne 1985, 1986; Sullivan 1986; Kaufmann 1987; Andrus et al. 1988) may be its primary contribution to the physical habitat of juvenile salmonids.

Reductions in LWD frequency and volume have been directly attributed to forest practices (Grette 1985; Heifetz et al. 1986; Lisle 1986; Murphy and Koski 1989; Bilby and Ward 1991). Logging can also change channel orientation and distribution of LWD in streams (Hogan 1986; Tripp and Poulin 1986a), often resulting in the accumulation of large, infrequently spaced debris jams (Bryant 1980; Bisson et al. 1987; Potts and Anderson 1990). Systematic logging of riparian areas over the last century has reduced the potential for LWD recruitment in many systems (Sedell et al. 1984; Andrus et al. 1988; Murphy and Koski 1989) and some of these streams may not recover completely for more than a century (Murphy and Koski 1989; Sedell et al. 1991). Recently, management techniques have been developed to reduce LWD loss and allow continued recruitment from riparian stands (Bilby and Wasserman 1989; Robison and Beschta 1990b).

In forested streams, LWD is associated with the majority of pools (Table 1) and the amount of LWD has a direct affect on pool volume, pool depth and the percentage of pool area in a stream (Elliot 1986; Murphy et al. 1986; Carlson et al. 1990; Beechie and Wyman 1992). The sediment storage capability of LWD (Triska and Cromack 1980; Sedell et al. 1984; Lisle 1986) is important in the temporal mitigation of rapid inputs of sediment from point sources such as landslides and channel failures (Bisson et al. 1987).

Some research has indicated that LWD frequency decreases with increasing channel width (Figure 6) while other studies show the opposite trend (Figure 7) (Tables 2a and 2b). Gradient and channel width tend to vary simultaneously and it is difficult to see a clear effect of gradient on LWD loading (Tables 3a and 3b). The average diameter, length and volume of individual pieces of wood increases with stream size (Bilby and Ward 1987, 1989, 1991; Robison and Beschta 1990a). As channel width increases, debris jams become larger, less frequent and more closely associated with the stream margins (Triska and Cromack 1980; Bisson et al. 1987).

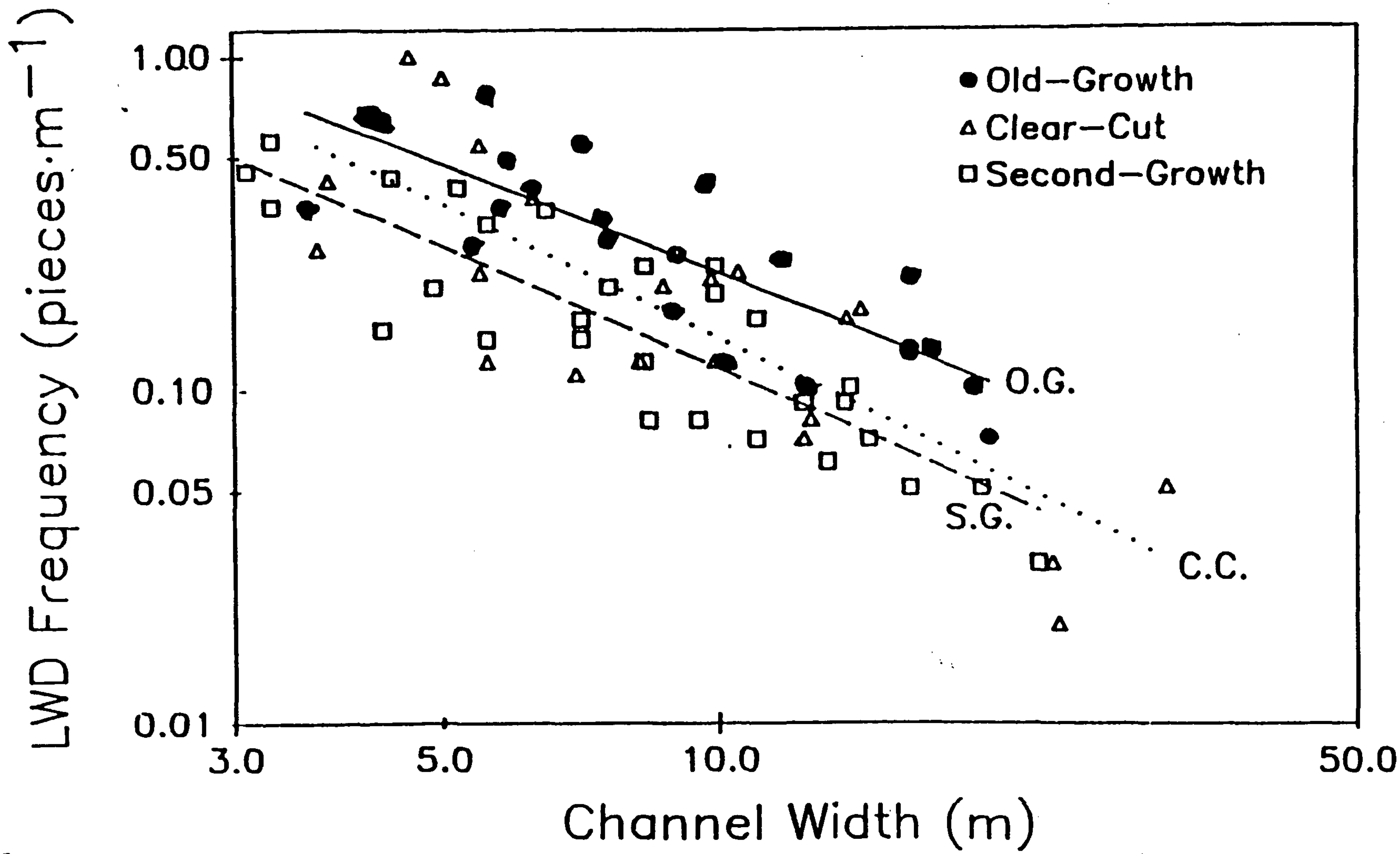
In some instances, LWD frequency is reduced in streams draining managed forests (Bilby and Ward 1991; Figure 6) and in others the difference is not detectable (Ralph et al. in preparation; Figure 7). However, due to recruitment from mature coniferous riparian stands, unmanaged forests have larger average debris sizes (Andrus et al. 1988; Ralph et al. in prep.; Figure 8). These larger pieces tend to be more stable (Lienkaemper and Swanson 1987) and do not decay as fast (Anderson et al. 1978). For these reasons debris volume may be an important indicator of the energy buffering capacity of streams and the effect of logging on this capacity. To detect the multiple impacts of forest practices on LWD loading, we use LWD frequency in pieces per channel width and debris volume index as units for the establishment of target conditions.

Target condition

Large woody debris frequency in streams draining unmanaged watersheds is highly variable (Tables 2a and 2b; Figure 7). One of the most complete data sets for unmanaged Washington streams is that of Bilby and Ward (1989) and target conditions have been set using their channel width dependent regression. A conversion of these data to pieces per channel width was performed to plainly display LWD loading for streams of different size. This conversion indicates that LWD frequency ranges from 2.44 pieces per channel width to 2.03 pieces per channel width as bank full width increases from 4m to 19m (Table 2a - column 1). Use of Bilby and Ward's (1989) regression equation to predict LWD loadings for streams of other sizes is not appropriate. Bilby and Wasserman (1989) indicate that LWD frequencies are similar in unmanaged sites in eastern Washington and other authors document similar or higher values in channels wider than 5m (Table 2a). The higher values of Robison and Beschta (1990a) and Murphy and Koski (1989) may, in part, be due to the use of a smaller minimum length criterion in these studies (1.5m and 1m respectively).

One notable inconsistency in the existing literature for unmanaged stream systems is the effect of channel width on LWD frequency (Tables 2a and 2b). Data from western Washington suggest a decrease in LWD frequency with increasing channel width (Bilby and Ward 1989, 1991; Figure 6), however, most other research in unmanaged systems demonstrates an opposite trend (Murphy and Koski 1989; Robison and Beschta 1990a; Fox 1992; Ralph et al. in prep.; Figure 7). While LWD size criteria differ between these studies (Table 4), both smaller and larger size criteria than those of Bilby and Ward (1989, 1991) were used. Bilby and Ward's

Figure 6: Relationship between numbers of pieces of LWD per length of surveyed channel and channel width for the three stand-age classes (O.G. = old-growth), C.C. = clear-cut, S. G. = second-growth). From Bilby and Ward (1991).



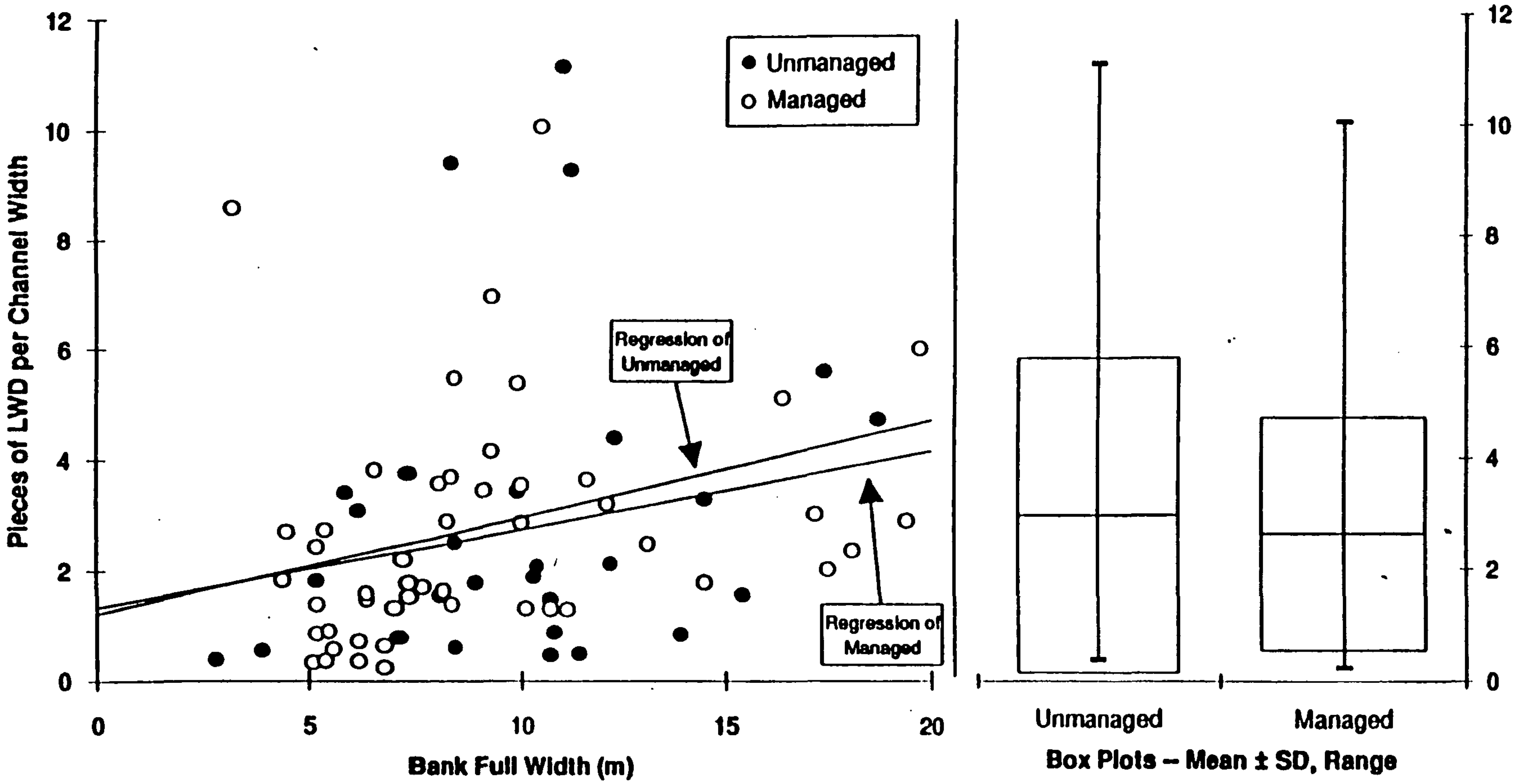


Figure 7: Relationship between number of pieces of LWD per channel width and channel width for managed and unmanaged streams of Washington. Mean values are not statistically different between managed and unmanaged streams. After Ralph et al. (in preparation) - 1990-1991 Ambient Monitoring data.

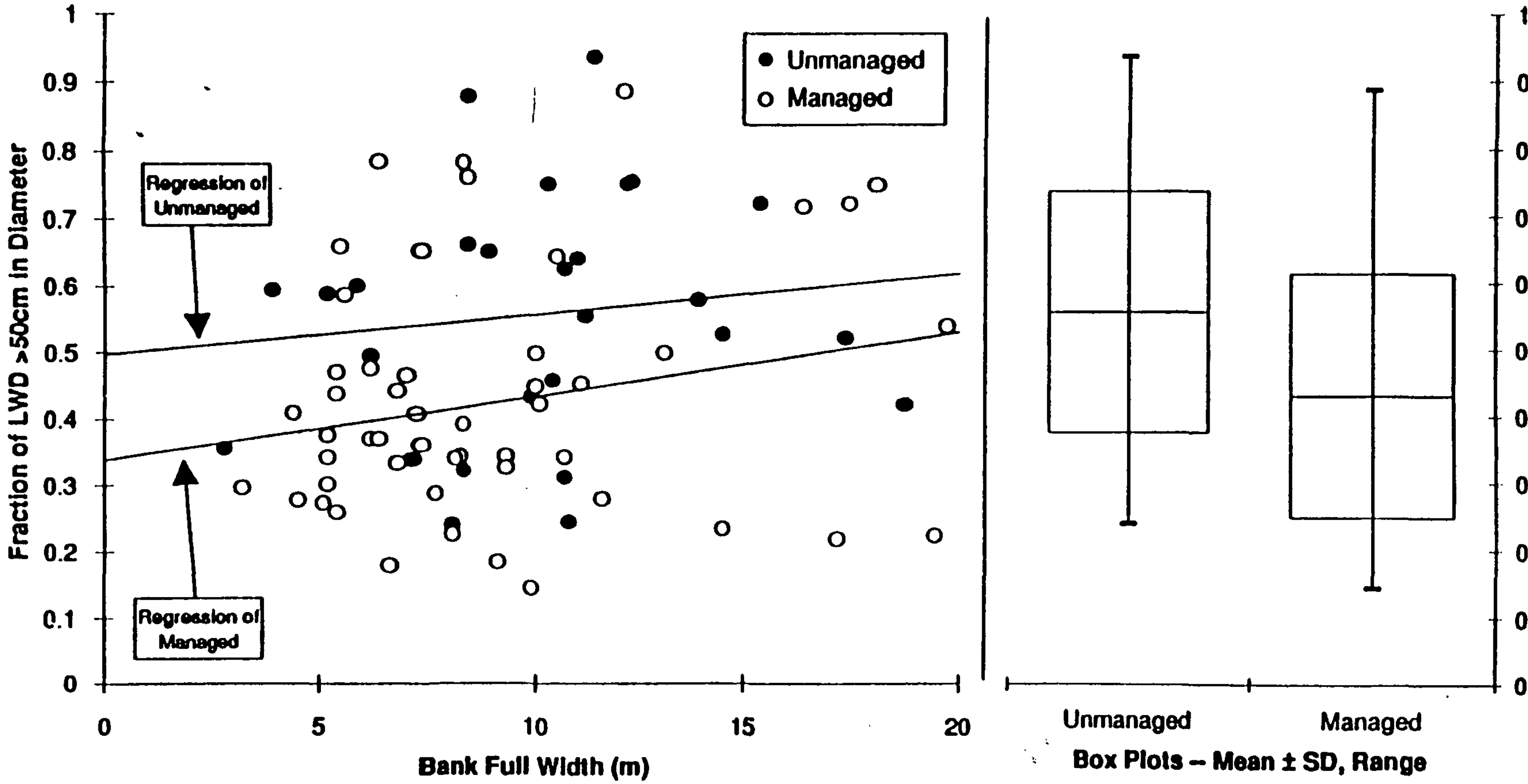


Figure 8: Relationship between the fraction of LWD pieces which are >50cm in diameter and channel width for managed and unmanaged streams of Washington. Mean values are statistically different between managed and unmanaged str^ams (p = 0.002). After Ralph et al. (in preparation) - 1990-1991 ıbient Monitoring survey data.

(1989) and Ralph et al.'s (in prep.) streams had a wide range of gradients (1-18%) while those examined in Alaska did not have gradients >3% (Murphy and Koski 1989; Robison and Beschta 1990a). Streams examined in Mt. Rainier National Forest also had a wide range of gradients (2-14%) (Fox 1992). Thus, the apparent inconsistency in LWD frequency trends by channel width cannot easily be explained and has yet to be thoroughly examined.

Average LWD frequencies from western Washington second-growth and clear-cut streams commonly fall below the target condition selected in this paper (Bilby and Ward 1991; Figure 6), however managed streams in other surveys often do not fall below this level (Table 5). This may be an indication of the need for further stratification or multiple indices which reflect frequency, piece size (see below) and species composition. Logging may not change the frequency of LWD pieces (Ralph et al. in prep.) but rather the relative size of these pieces (Figure 8). Therefore, a target condition which reflects the average size of LWD pieces will be useful in detecting the cumulative effects of logging. For the determination of a target condition based on LWD piece size, we have used Bilby and Ward's (1989) regression for unmanaged streams of western Washington (Table 6 - column 1). Second-growth streams >7m wide in western Washington and the Stillaguamish and Snohomish River basins had a debris volume index significantly less than the target condition (Bilby and Ward 1991; Table 6).

Methodology

The size criteria (>10cm diameter) so ubiquitously applied to the definition of LWD was originally based on the ease of handling logging slash during stream cleaning operations (Froehlich 1973). Since this time, criteria used to define LWD have varied considerably between studies (Table 4). Unstable (loose) pieces of LWD are commonly excluded from stream surveys (Bilby and Ward 1989), however the determination of stability is somewhat subjective (Lienkaemper and Swanson 1987). In the absence definitive studies which show the relative role of woody debris with different characteristics, we recommend the continued use of >10cm in diameter and >1m in length as size criteria. This will allow comparison of data to existing studies and additional analyses can be performed for larger size classes. As the target conditions specified in this paper were determined for LWD >2m long, this length criterion must be used in the comparison of survey results to target conditions.

To maximize the information in an LWD survey, we recommend the methods of Robison and Beschta (1990a). Exclusion of any of the attributes of their methods reduces a survey's ability to make conclusions about existing conditions and future prospects for debris loading. Basin wide inventories of LWD in fish bearing and non-fish bearing channels are essential to establish the variability caused by local anomalies and catastrophic disturbances. Without basin-wide inventories it will be impossible to develop a solution for the recovery of LWD levels.

To relate an LWD piece count to the target condition specified in this report, the number of LWD pieces in a stream reach should be divided by the length of that reach in meters and multiplied by the average bank full channel width in meters. Debris volume index should be calculated using the methods of Bilby and Ward (1989). Debris volume index values and LWD frequencies greater than those specified in this paper indicate a condition which is similar to that of unmanaged streams (target condition).

Recommendations and Conclusions

1. The target condition for LWD frequency is based on Bilby and Ward's (1989) channel-width dependent regression for unmanaged streams in western Washington. Debris frequencies derived from this regression have been multiplied by channel width to present specific LWD target values for a range of stream sizes (Table 2a - column 1).

2. The target conditions for debris volume index are those predicted by Bilby and Ward's (1989) channel-width dependent regression for unmanaged streams of western Washington (Table 6 - column 1).

3. Target conditions for LWD frequency and debris volume index apply statewide for streams 4m to 19m bank full channel width.

4. For the comparison of LWD surveys to target conditions, a size criteria of >10cm diameter and >2m length should be used.

5. Large woody debris surveys should be done on a basin-wide scale using the techniques of Robison and Beschta (1990a) and size criteria of >10cm in diameter and >1m in length. Surveys should not be limited only to fish bearing reaches.

Pools

Many species and age-classes of juvenile salmonids are dependent on pools for rearing habitat (Bustard and Narver 1975a: Tschaplinski and Hartman 1983; Heifetz et al. 1986; Chisholm et al. 1987; Bugert and Bjornn 1991; Heggenes et al. 1991a, 1991b). The abundance of juvenile coho in particular appears to be more strongly influenced by the amount and quality of available pool habitat than other variables, including instream debris (Nickelson et al. 1979; Carman et al. 1984; Murphy et al. 1986). Different species and age classes of salmonids prefer pools with different characteristics. Coho are abundant in all pool types, age-0 salmon and trout are common in dammed and plunge pools, and older trout are increasingly common in scour pools (Bisson et al. 1982; Bisson et al. 1988). Due to their increased depth, pools are directly beneficial to salmonids through the provision of shelter from predators and refuge during summer low flow periods (Bugert et al. 1991; Heggenes et al. 1991b). In association with local obstructions, pools also provide areas of reduced velocity which are used by juveniles while rearing (Bisson et al. 1988; Shirvell 1990; Heggenes et al. 1991b) and adults while migrating and spawning (Bjornn and Reiser 1991).

The formation and maintenance of pools in streams is an inherent product of fluid dynamics and is largely independent of channel bank and stream bed material (Keller and Melhorn 1978). Pool frequency appears to be related to gradient, channel width and, in small forested

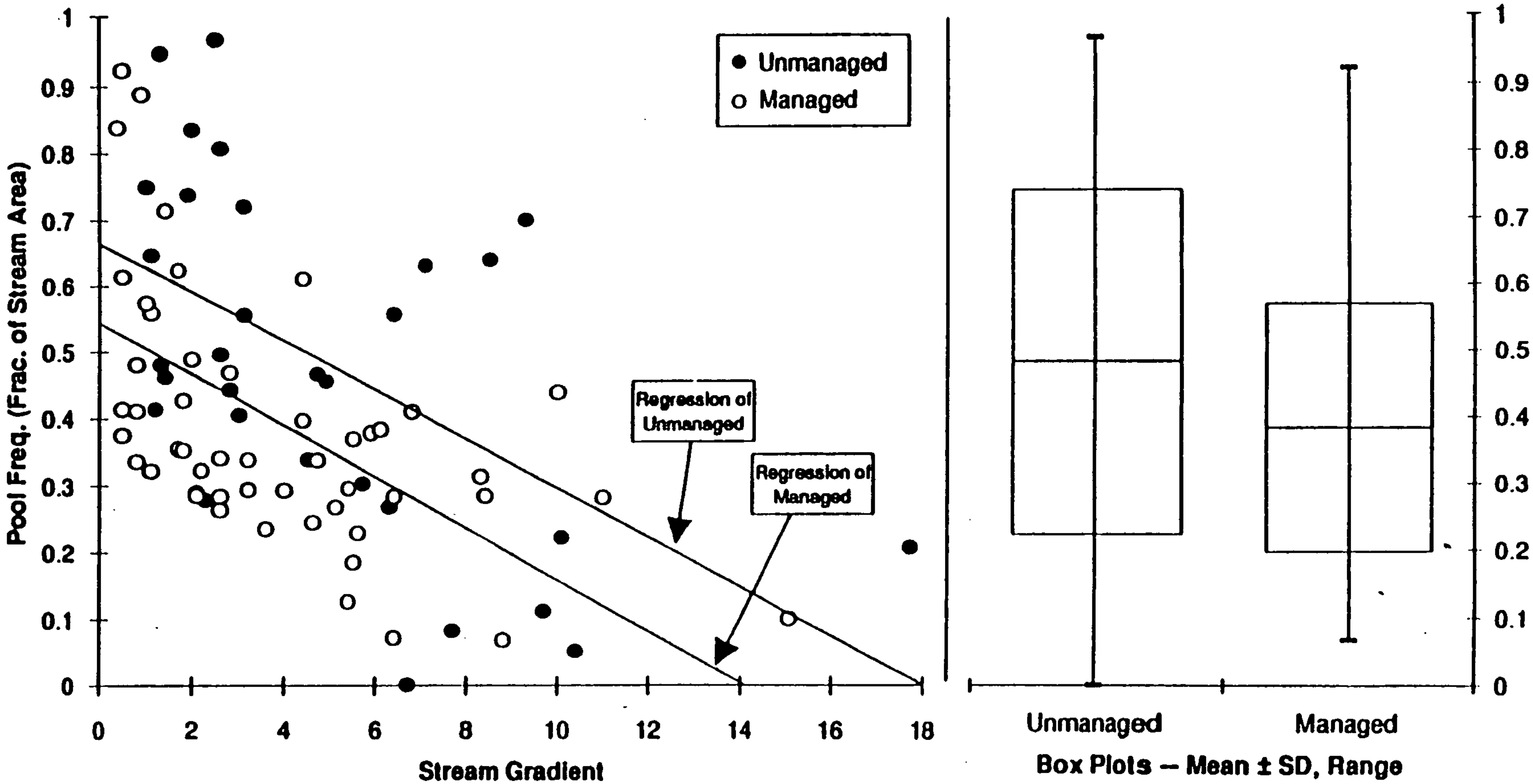


Figure 9: Relationship between the fraction of the stream area comprised of pools (% Pools) and stream gradient for managed and unmanaged streams of Washington. Using an unbalanced ANOVA to factor out the effect of gradient, managed was found to have a statistically significant effect ($p = 0.004$). After Ralph et al. (in preparation) - 1990-1991 Ambient Monitoring survey data.

streams, local obstructions (Bilby and Ward 1989, 1991). As stream size increases (>7m), the relative role of LWD in determining pool surface area increases (Bilby and Ward 1989) and as gradient increases, pools become spaced at shorter intervals in step-pool formations (Chin 1990; Grant et al. 1990).

Streams draining managed forests often have reduced pool frequencies, volumes, surface areas, and average depths (Bjornn et al. 1977; Lisle 1982; Hogan 1986; Tripp and Poulin 1986b; Sedell et al. 1990; Bilby and Ward 1991; Ralph et al. in prep.). This loss of pool habitat is often the direct result of a decrease in LWD (Bisson and Sedell 1984; Elliot 1986) and an increase in sediment load (Bjornn et al. 1977; Lisle 1982; Everest et al. 1987; Lisle and Hilton 1991). Woody debris creates complex pool types such as plunge and dammed pools (Bisson et al. 1982) and correspondingly, streams with less wood have an increased proportion of scour pools (Bilby and Ward 1991; Ralph et al. 1991).

Indices of pool abundance and character vary considerably between studies (Table 7). Of these indices, percent pools (percentage of the wetted surface area which is comprised of pools) is the most common. This measure shows a strong positive correlation to coho parr total densities (Murphy 1983; Carman et al. 1984) and decreases as a result of debris torrents and clear-cutting (Heifetz et al. 1986; Johnson et al. 1986; Tripp and Poulin 1986b). Total pool volume is also related to fish abundance and forest practices, however, data for this index in unmanaged forest is not as extensive. "Primary pool" is a narrow definition of pools that one would expect to find in a low gradient meandering channel form on the outside of each meander bend (Keller and Melhorn 1973, 1978). As such it is expected that this parameter would be relatively insensitive to management influences except in extreme cases and should not be used to describe desirable conditions.

Target condition

Percent pools and pool frequency vary considerably in surveys of streams draining old growth forests (Tables 8a, 8b and 8c; Figure 9) and some of the variation in these values is likely due to variation in pool identification criteria (Table 9). Studies using the habitat classification system of Bisson et al. (1982) indicate percent pools in unmanaged streams usually ranges from 39% to 67% (Table 8a). The lower pool percentages of Tripp and Poulin (1986b) and Carlson et al. (1990) may be an artifact of different pool identification criteria (Table 9) and the higher values of Grette (1985) are likely due to the inclusion of runs and glides in pool percentage calculations.

Based on data collected using the closely related pool classification techniques of Bisson et al. (1982) and Sullivan (1986), a target condition of 50% pools is generally indicative of pool habitat observed in streams with gradients < 3% draining unmanaged forests (Table 8a). Pool percentage decreases with increasing channel gradient (Table 8a; Figure 9; Beechie unpublished data) but data are insufficient at this time to establish a target condition for steeper streams (>3%). Further analysis of stream pool percentages by gradient in the 1991 Ambient Monitoring data base may allow the determination of target conditions for streams >3%.

Surveys of managed streams indicate that pool percentage falls below 50% in the majority of impacted basins (Heifetz et al. 1986; Beechie and Sibley 1990b; Ralph et al. 1991; Figure 9). Of surveyed streams within the Stillaguamish and Snohomish River systems having gradients <3%, 9 would meet a target condition of >50% pools while 13 would fall below this level (Beechie unpublished data). In this survey, 3 managed streams with gradients >3% all had <30% pools. Percent pools in 4 managed Skagit River watersheds ranged from 9 to 32 (Beechie and Wyman 1992).

Methodology

Criteria used to identify pools vary widely in the scientific literature (Table 9). The pool classification system of Bisson et al. (1982) appears to be sensitive to management response (Bilby and Ward 1991) and relates well to preferential use by different species and age classes of salmonids (Bisson et al. 1988). This system had been widely accepted and we recommend its continued use in monitoring programs designed to assess the cumulative effects of forest management.

Surveys of pool habitat should specify the relative area of different pool types as well as overall pool percentage. As some measure of pool volume could also form the basis for additional target conditions, surveys should include residual pool depth (Lisle 1987) as well as surface area dimensions. Due to the role channel width and gradient play in pool maintenance and character, these factors should be noted in association with each pool. Pool area depends on stream stage (Beechie and Sibley 1990b) and in order to standardize measurements, surveys should be conducted during summer low flow periods.

Recommendations

1. The target condition for the percentage of the stream surface area comprised of pools is 50%. This target condition applies only to streams with gradients <3%.

2. Limited data and large variability have not allowed precise determinations of pool percentage target conditions for streams with gradients >3%. However, we believe that the analysis presented in Figure 9, using data from Ralph et al. (in preparation), may form the basis for specific target conditions in streams with gradients between 1% and 18%.

3. Measurements of pool area should be made at summer low flow and follow the convention of Bisson et al. (1982).

4. While percent pools is used in the specification of a target condition, average pool volume or total pool volume by channel width and gradient may provide a more sensitive indicator of cumulative effects. Residual pool depth (Lisle 1986) may form a good surrogate measure of pool volume.

Substrate Composition

Studies of the effects of fine sediment on the reproductive success of salmonids present a wide array of methodologies, textural indices of substrate composition and conclusions (Kondolf 1988; Young et al. 1991a; Tables 10-13). In general, salmonid survival-to-emergence (STE) decreases as the amount of small particles in the substrate increases (Lotspeich and Everest 1981; Shirazi and Seim 1981; Tappel and Bjornn 1983; Chapman 1988; Young et al. 1991a). These 'smaller' sediment sizes cause reduced STE by entrapping alevins within the streambed and limiting inter-gravel flow of oxygenated water to developing embryos, which reduces dissolved oxygen levels and concentrates embryo waste products (Kondolf 1988; Bjornn and Reiser 1991). Although spawning salmonids remove smaller sediments from their egg pockets and redds (Everest et al. 1982; Young et al. 1989), the amount of fines in the egg pocket is related to the amount of fines in surrounding gravels (Grost et al. 1991a). Fines continue to accumulate in the surface layer and intrude into the redd during the incubation period (Beschta and Jackson 1979; Grost et al. 1991).

A variety of forest practices are known to significantly increase sediment delivery to streams (Everest et al. 1987). In some basins, the road system is a primary contributor (Reid 1980; Cederholm et al. 1981; Reid and Dunne 1984; Wasserman 1988) while in others slope failures and stream bank erosion are most influential (Roberts 1987; Schlichte et al. 1991; Scrivener and Brownlee 1989). Stream surveys have detected increases in fine sediment levels with increased logging activity and logging road density (Cederholm et al. 1981; Wasserman 1988; Scrivener and Brownlee 1989). A moratorium on logging in the South Fork Salmon River Basin resulted in a considerable decrease in fine sediment levels in spawning gravels (Platts et al. 1989). Basin geology is another factor that can affect the amount of fine sediment in streambed gravels (Duncan and Ward 1985) and different basins often have different substrate compositions (Figure 10).

All of the indices used to specify gravel composition have merits and limitations. Recent reviewers have concluded that a single measure of substrate composition is probably inadequate to index both salmonid survival-to-emergence and management induced textural changes (Chapman 1988; Young et al. 1991a; Scrivener in review). Scrivener (in review) notes that the selection of a method for determining changes in gravel composition should be based in part on watershed sediment characteristics. The most effective monitoring approach would be to focus analysis on that portion of the particle size distribution that management activities are influencing.

Recent analysis indicates that the percentage of fines less than a certain size alone may not be the best predictor of salmonid survival to emergence (see Table 2 in Young et al. 1991a). One limitation of the percent fines index is that more than one size fraction is damaging to incubating and emerging salmonids (Table 12). In fact, STE can vary dramatically at a fixed amount of fines less than one size while the amount of other size fractions changes (Tappel and

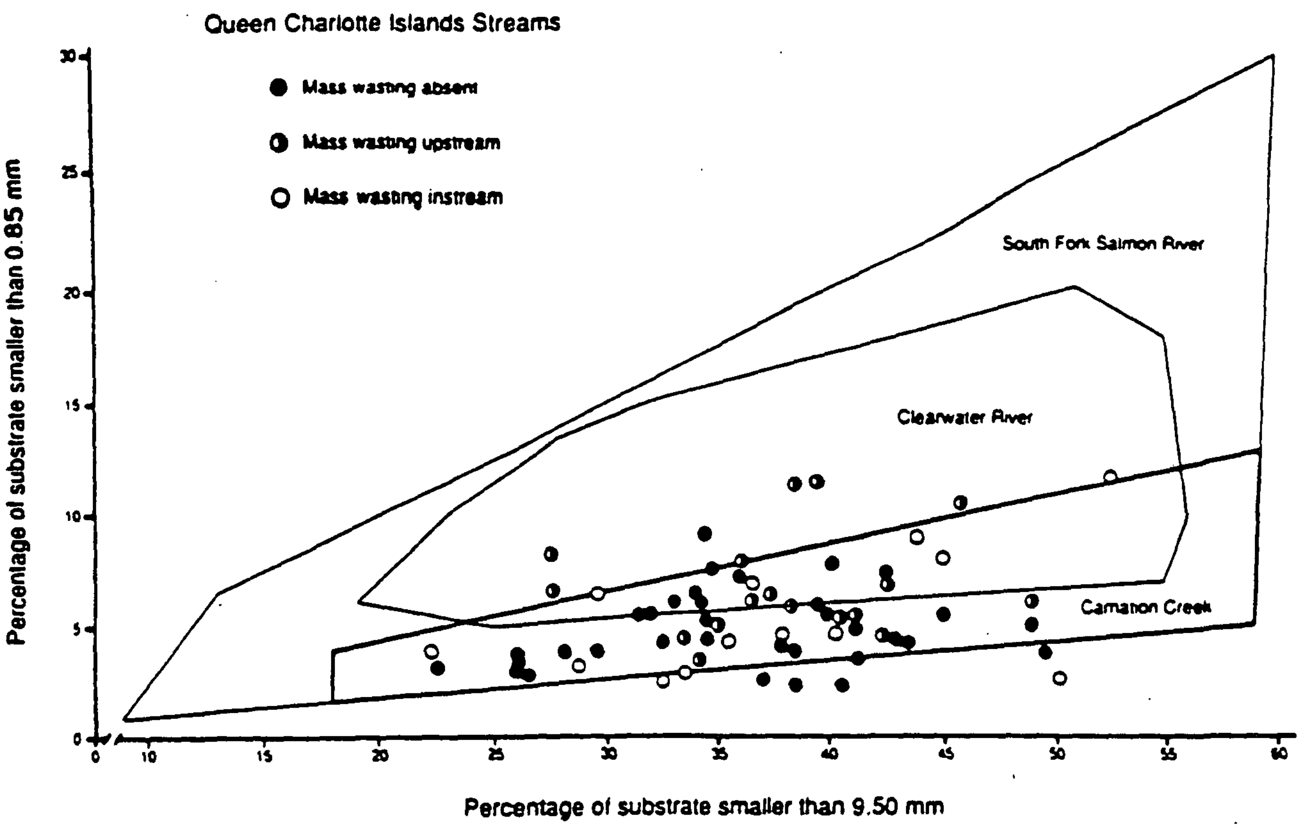


Figure 10: Spawning gravel composition in streams of the Queen Charlotte Islands (Tripp and Poulin 1986a), South Fork Salmon Salmon River (Tappel and Bjornn 1983), Clearwater River (Tappel and Bjornn 1983), and Carnation Creek (Scrivener and Brownlee 1987). From Tripp and Poulin (1986a).

Bjornn 1983; Irving and Bjornn 1984). In addition, the use of a single size criterion will not detect changes in substrate composition which occur outside that specific size fraction. Equation graphics (Tappel and Bjornn 1983) was designed to counter this problem by incorporating the amount of fines less than the two substrate sizes which best predict overall substrate composition. However, very few studies have attempted to apply this technique.

Geometric mean particle size (Shirazi and Seim 1981) and the fredle index (Lotspeich and Everest 1981) are methods used to express substrate composition which reflect the entire particle size distribution. Both of these measures of central tendency are strongly related to STE but appear less precise than percent fines in detecting changes in natural streambed composition (Young et al. 1991a). However, percent fines is only more accurate in the later function if the size criterion correctly targets that portion of the particle size distribution that is changing. Both geometric mean particle size and the fredle index have detractors and benefactors and may actually be equally accurate in predicting STE (Young et al. 1990; Table 11). However, the fredle index performs better in detecting changes in natural streambed gravels and is probably accurate in a wider range of applications (Scrivener in review).

Although percent fines less than a certain size may not be the most desirable way to specify substrate composition for predicting STE, the majority of information on gravel surveys in unmanaged streams is specified in this manner. This size fraction has also been identified as one of the most detrimental to salmonid STE (Table 12) and many studies have used this criterion (Tables 13 and 14). Target conditions are specified using percent fines <0.85mm in diameter since the majority of data sets draining unmanaged forests are expressed in this manner (Table 14).

Target condition

In the following discussion of substrate composition, 'fines' refers to the percentage of the substrate which is comprised of particles <0.85mm in diameter. Survival-to-emergence at a specific amount of fines varies considerably (Table 10) and a criterion based on these studies would arbitrarily choose a single definitive study. Since we have chosen the ecosystem approach for setting target conditions, this does not present a problem. Cederholm and Reid (1987) indicate that levels of fines in unmanaged streams in the Olympic National Forest averaged 6.37% and Hatten (1991) found an average of 10.86% in unaffected Hoh River tributaries. Surveys of conditions in the South Fork Hoh and Main Hoh Rivers detected levels of fines in undisturbed areas of 11.38% and 14.5% respectively (Cederholm 1991). Levels of fines in southeast Alaska averaged 9.45% (Edington 1984) and 9.65% (Sheridan et al. 1984). Values observed in streams draining unmanaged forests are presented in Table 14. We have chosen a target condition of 11% fines <0.85mm as this appears to represent the level around which a majority of the sites in Washington cluster. The 11% target condition should be applied to low and moderate gradient streams (<3%) up to 30 meters in channel width.

Basin geology can have a significant effect on percent fines and this suggests that a universal target condition applied indiscriminately across geologic boundaries may be inappropriate. For example, levels of fines in some unmanaged coastal Oregon streams are considerably higher than those in Washington (Koski 1966; Adams and Beschta 1980; Ringler and Hall 1988). In contrast, average fines <1.19mm did not exceed 8% in Carnation Creek (Scrivener and Brownlee 1989) and values in the Queen Charlotte Islands are similar to those of Carnation Creek (Tripp and Poulin 1986a; Figure 10).

We have elected not to set a poor threshold for fine sediment and it is inappropriate to set additional target conditions just because it would be convenient. As with other parameters it would merely be an arbitrary decision in light of the diverse settings where the target condition would be applied. We have reexamined several surveys of managed streams and the number of streams not meeting a target level of 11% fines have been noted in Table 15. We recognize there will be substantial variability and a good number of managed streams will likely be in the 11-16% range for percent fines. Rather than blithely rely on a second threshold to denote poor conditions, the most appropriate management response is to thoroughly investigate possible reasons for fine sediment concentrations >11%.

Methodology

Sampling protocols designed to compare streambed substrate condition to a target condition requires a compromise between convenience and precision. Gravel composition in salmonid egg pockets and redds differs from that in general spawning reaches and this composition is modified during spawning (Everest et al. 1987; Chapman 1988). However, the amount of fines in egg pockets and redds appears to be correlated to the amount of these particles in the surrounding substrate (Grost et al. 1991a). Bulk sediment samples are only an index to the background substrate composition and actual conditions to which the eggs are exposed during incubation vary. Of the methods available for bulk substrate sampling, the McNeil cylinder yields minimal bias and is applicable to the widest range of sampling situations (Grost et al. 1991b; Young et al. 1991b).

Considerable subjectivity is often injected into substrate sampling due to the lack of systematic positioning of sample location. To standardize site selection, sampling should take place before spawning on transects in several spawning reaches. A standard sampling protocol which provides information representative of a reach or stream has not been satisfactorily described. A study which systematically determines these protocols based on statistical variability should be undertaken. Substrate composition analyses should pass sediment through a series of geometrically smaller sieves and the amount retained on each sieve should be measured using the wet volumetric or dry gravimetric methods (see Everest et al. 1982). An exacting comparison of these two methods is a research priority. To provide a thorough representation of data from field surveys and STE experiments, the specification of substrate composition should include all available indices (see Young et al. 1991a for methods to express substrate composition).

Recommendations and conclusions

1. A target condition for fines <0.85mm is set at 11%. Although there are some documented cases in unmanaged streams of higher levels, the majority of cases fall below this level. This target condition applies broadly to streams of different sizes and gradients but as a general rule would be most applicable to streams <3% gradient and between 5 and 30 meters bank full channel width.

2. Ideally, fine sediment target conditions should be based on natural levels observed at local undisturbed sites with similar characteristics.

3. A single method for the specification of gravel quality is inadequate to predict both salmonid survival-to-emergence and management induced changes in substrate composition. Future studies should specify substrate composition using a variety of indices (Young et al. 1991a).

4. Substrate should be sampled in potential spawning reaches prior to spawning using a McNeil cylinder and processed using standard methodologies (Everest et al. 1982).

5. Standard substrate sampling and processing protocols need to be established to ensure reliability of data being collected statewide.

Gravel Stability (Scour)

Salmonids have evolved elaborate behavioral life history adaptations that ensure their survival in highly dynamic stream environments. Many of these adaptations rely on the mobility of juvenile stages. However, during their embryonic development in the intra-gravel environment, they are not mobile and must rely in part on the stability of the streambed for their survival. Site selection and preparation of the spawning nest by the adults appears to maximize the survival of incubating eggs and alevins (Chapman 1988; Young et al. 1989; Grost et al. 1991). Egg burial depth is positively correlated to fish size and protection from physical disturbance during scour events (Ottaway et al. 1981; van den Berghe and Gross 1984; Crisp and Carling 1989).

Fluvial processes in gravel bedded riverine systems can have profound effects on the reproductive success of salmonids. During high flows that approximate or exceed bank full discharge, steambed gravels are mobilized causing scour (Andrews 1983). Common scour depths (Table 16) often encompass a large portion of the range of salmonid egg burial depths (Table 17) and dislodgment of eggs can be considerable (Figure 11). In studies of chum and pink salmon incubation conditions, McNeil (1966, 1969) concluded that while scour varied annually, egg losses frequently exceed those that could be expected based on fine sediment concentrations. In a comparison of streams affected and unaffected by debris torrents, Tripp and Poulin (1986a) estimated egg scour rates as high as 80-90% and concluded that in some highly disturbed streams, egg loss resulting from scour could be a constant and long term problem overshadowing the effects of fine sediments. Researchers in southwest Oregon have concluded

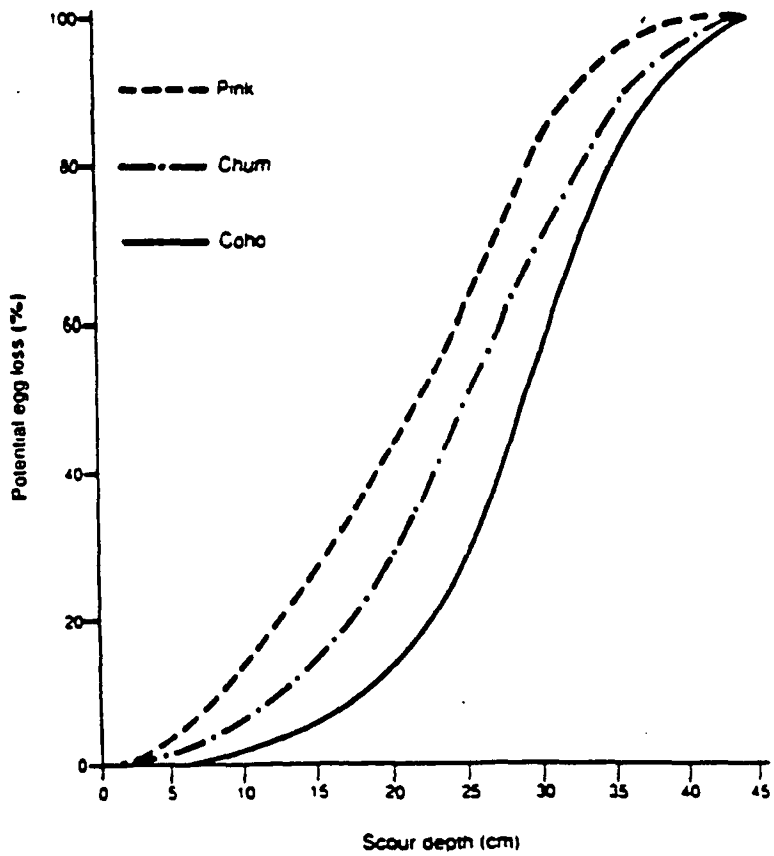


Figure 11. Relationship between egg loss and scour depth for pink, chum and coho salmon on the Queen Charlotte Islands. Curves for chum and coho salmon are based on a mean egg depth of 20 cm (from Klassen 1984). From Tripp and Poulin (1986a).

that streambed stability is a significant factor affecting the persistence of chinook stocks in that region and determined that 75% of the redds they monitored were either scoured or buried to damaging depths (Nawa et al. 1990). Scour can also expose deeper gravel layers to the intrusion of fines which adversely affect survival-to-emergence (Beschta and Jackson 1979; Lisle 1989).

Disturbance of natural streambed stability is tied to changes in sediment and flow regimes within a basin (Reid et al. 1985; Kondolf et al. 1991). Logging affects both these factors (Scrivener and Brownlee 1989) and their combination is perhaps the most damaging. Increases in the frequency of channel forming flows (those discharges likely to cause scour and fill) are especially damaging because they expose the stock to mortality schedules under which they did not evolve. It is likely that mortality rates may be increased to levels beyond which natural fecundity or multiple age maturing life history characteristics can compensate.

Target condition

The greatest limitation on the use of scour to describe gravel stability is the large degree of spatial and temporal variability (Lisle 1989; Table 16). Variability within basins and between basins with similar precipitation patterns may be due to local obstructions and streambed composition (Kaufmann 1987; Scrivener and Brownlee 1989). In order to compensate for inherent variability, sampling must be extensive, monitoring frequent and values averaged over several years. We do not recommend that gravel stability be used as a parameter for the detection of cumulative effects of logging within the current proposed monitoring system. However, a known depth of scour bears a direct relationship to the removal of eggs buried at known depths (Tripp and Poulin 1986a; Figure 11). Therefore gravel stability may be a useful indicator of egg loss due to high stream flows and the cumulative effects of logging if implemented in an intensive monitoring program. Establishment of specific target conditions for gravel stability could be determined through the observations of average scour depths in unmanaged streams. Caution should be exercised in the extrapolation of scour depth to egg loss predictions as egg burial depth is dependent on species and female size (van den Berghe and Gross 1984; Crisp and Carling 1988).

Methodology

Depth of scour and fill can be determined using a number of simple field tools such as sliding-bead monitors, ping pong or golf ball monitors, cross-sectional surveys and stand-pipes (Duncan and Ward 1985; Lisle and Eads 1991; Nawa and Frissell 1991). In a unique study done for the Seattle Water Department (1991) researchers used buried radio transmitters at specific depths to relate discharge to specific scour depths. The percentage of eroded stream banks correlates well to the extent of streambed instability in southwest Oregon and may be a valuable surrogate measure for streambed stability (Chris Frissell personal communication). Currently, we recommend the golf ball monitor (Tripp and Poulin 1986a) to monitor scour in salmonid streams.

Recommendations and Conclusions

1. Depth of scour can critically affect long term viability of salmonid stocks and is frequently more severe in intensively managed systems.

2. The intensive nature of sampling required for the accurate and precise determination of gravel scour depth precludes its present use in a widespread monitoring program.

3. An acceptable field technique for the measurement of scour in salmonid streams is the use of plastic golf ball monitors (Tripp and Poulin 1986a). Percent streambank erosion may be a reasonable surrogate variable for streambed instability (Frissell pers. comm.). Additional research needs to be done to develop protocols for use in a widespread monitoring program.

Interstitial Space

Substrate "roughness" and interstitial space are important factors influencing the rearing densities of several species of salmonids. Steelhead, Atlantic salmon, chinook, cutthroat, brook trout and brown trout have all been shown to prefer stream substrates with interstitial voids (see review in Chapman and McLeod 1987). Age 0 Atlantic salmon and steelhead often seek refuge within the substrate especially during low winter stream temperatures (Hartman 1963, 1965; Everest 1969; Kelley and Dettman 1980; Rimmer et al. 1981; Cunjak 1988). Experimentally induced reduction of interstitial voids with coarse sands in a natural stream has resulted in reduced juvenile salmonid densities (Alexander and Hansen 1983).

Although the general value of "interstitial space" is well established for several salmonids, especially the young-of-the-year age class, quantification of this parameter and its use as an indicator of management impacts, is currently restricted to Idaho. We believe this situation is the result of a combination of two factors; 1) interstitial space appears to be an especially important local variable considering the snow melt dominated hydrograph, cold winter water temperatures and demonstrated winter behavior of juvenile salmonids to hide in the substrate and 2) coarse granitic sands produced by erosion in watersheds draining the Idaho batholith are especially prone to filling streambed interstices.

In the rain dominated hydrographic regions of Washington, it is not clear that hiding in the substrate would be an advantageous behavior for juvenile salmonids since the streambed gravels are often mobilized by high flow events. Rather, evidence from the Olympic Peninsula suggests that young-of-the-year trout seek refuge in small runoff tributaries (Cederholm and Scarlett 1982). In some areas of eastern Washington with snow melt dominated hydrographs and cold winter water temperatures, substrate interstitial space may be a favored winter refuge for some salmonids, however, no information exists for this area.

Management activities that increase fine sediments are well known (Cederholm et al. 1981; Reid and Dunne 1984; Scrivener and Brownlee 1989). Several studies have concluded that local inputs of fine sediment reduce interstitial space (Burns 1984; Potyondy 1988; Ries et al. 1991). However, direct correlations between forest management and interstitial space or percent cobble embeddedness (PCE) have not been established. This may, in large part, be due to the substantial spatial and temporal variation of these variables (Potyondy 1988, 1991; Parker et al. 1989; Ries et al. 1991).

Target condition

Before interstitial space target conditions could be set in Washington, this habitat feature would have to be demonstrated to have biological significance to salmonids of the region. A systematic examination of the importance of this attribute by species and hydrographic condition is desirable. Baseline inventories should be conducted to establish natural levels of interstitial space in all areas of Washington. These surveys should initially be focused on streams not influenced by management activities.

Methodology

Measurements of interstitial space can be expressed using percent cobble embeddedness (PCE) (Burns and Edwards 1985; Burton and Harvey 1990), ocular embeddedness (Torquemada and Platts 1988), number of free-matrix particles (Ries et al. 1991) and interstitial space index (ISI) (Vadeboncoeur 1988). Of these indices, ISI is the most sensitive to change and bears the closest relationship to the habitat requirements of juvenile salmonids (Vadeboncoeur 1988; MacDonald et al. 1991). The specification of mean particle size in conjunction with ISI may provide a valuable indication of the relative complexity of available interstitial space.

Measurements of ISI (or PCE) can be made at random sites (Burns 1984) or on transects (Skille and King 1989) and each rock can be treated as a single measurement or values can be averaged for a hoop (Skille and King 1989). Corrections for samples with >10% sand are also useful (Torquemada and Platts 1988). If studies are undertaken to categorize interstitial space in the state of Washington, we recommend the use of the transect methods of Skille and King (1989) and that values be specified in ISI units (Vadeboncoeur and Kramer 1988). Due observed variability, Ries and Burns (1989) suggest that data must be collected over a minimum of five years in order to adequately describe interstitial space conditions and trends.

Measurements of interstitial space were developed for watersheds with coarse granitic sand and are not appropriate in basaltic streams (Burns and Edwards 1985; Potyondy 1988). Due to limitations of sampling methods, embeddedness and ISI measurements are best applied to streams >6m wide (Potyondy 1988) with gradients <3% (Skille and King 1989).

Other methods which monitor the levels of fine sediment in a stream may also serve as good indicators of a loss in interstitial space. In addition, other measures of textural composition of the armor layer may provide accurate indexes of interstitial space. A relatively new technique which measures the percent of the residual pool volume filled with fine sediments (Lisle and Hilton 1991), may be a good indicator of the effects of fine sediments on a variety of important habitat components. We believe this method has promise as an index for: 1) the loss of pool rearing space due to filling with fine sediments, 2) the amount of fine sediments likely to be mobilized, transported across and entrained in freshly spawned areas of the stream and 3) the loss of interstitial voids in the substrate.

Recommendations and Conclusions

1. Data on interstitial space in the state of Washington is lacking. We do not currently recommend interstitial space as a monitoring parameter and no target condition can be set.

2. Interstitial space is an important component of juvenile steelhead rearing habitat and is also beneficial to other species. Future research in Washington is warranted, but must be specific to hydrographic region.

3. Monitoring of interstitial space or cobble embeddedness has the greatest application in low gradient streams (<3%) draining watersheds with weathered granitic geology.

4. Interstitial Space Index (ISI) is more sensitive to change and bears a closer relationship to salmonid habitat requirements than Percent Cobble Embeddedness (PCE). Future research should specify values in ISI. Specification of mean particle size may be a valuable supplement to the categorization of interstitial space.

5. ISI measurements should be made using the techniques of Vadeboncoeur and Kramer (1988) and the sampling design Skille and King (1989).

ADDITIONAL RESEARCH NEEDS

1. A system of stream classification that will allow appropriate stratification of streams for the purposes of assigning target conditions for a wide range of parameters.

2. Surveys of habitat conditions in streams draining unmanaged forests of differing geomorphic, climatic and geologic settings. These surveys must be done in a systematic manner and fit into a pre-defined stratification system. If they are not conducted in this manner their utility to a target conditions approach to channel evaluation will be limited. These surveys should be done on a basin scale and include but not be limited to the parameters identified in this report. Ralph et al. (1991) and MacDonald (1991) should be consulted for information on conducting quality, replicable stream surveys.

3. Surveys of streams in managed forests should be conducted for comparison purposes. These surveys should be conducted in basins that have been stratified by natural watershed sensitivities (soils, slopes and elevations) and the level of forest management (stand age distribution and road density). The Olympic National Forest has already stratified their watersheds in this manner on a GIS and would serve as a good template. This study design may be a simple yet elegant way to begin collecting data to corroborate or disprove some of the existing notions about CEs.

4. Other parameters that would fit well into the ecosystem approach of setting target conditions are measures of channel geometry and basin hydrology. Orsborn (1990) should be consulted for these concepts. A good example of their application to the assessment of channel condition is Madej (1978).

5. Standardized sampling protocols are needed for a variety of methodologies.

a. Protocols for substrate sampling using a McNeil cylinder which produce values representative of a spawning area, reach, and basin need to be systematically determined.

b. A comparison of the merits of wet volumetric and dry gravimetric methods for the processing of substrate samples needs to be made in order to establish a standardized method.

c. A definition of large woody debris which is based on its ability to influence channel morphology and fish abundance would be a welcome addition.

6. Several methodologies may form the basis for rapid and reliable surrogate measures of more detailed and time consuming instream parameters.

a. Percent pool filling with fine sediments (Lisle and Hilton 1991) may form a surrogate measure for the influence of sedimentation on pool loss, spawning substrate composition and interstitial space. The use of these technique as a surrogate for these variables needs to be examined. Values for unmanaged streams in Washington need to be established.

b. Percent bank erosion may form a surrogate measure for streambed stability and would be widely applicable in a widespread monitoring program (Chris Frissell pers. comm.).

TABLES

Table 1: Relative contribution of LWD to pool formation. Criteria used to define LWD (Table 4) and pools (Table 9) vary between studies.

AUTHOR	DATE	% OF POOLS FORMED BY LWD	LOCATION
Andrus et al.	1988	70	coastal Oregon
Bilby	1984	80	southwest Washington
Carlson et al.	1990	64	northeast Oregon
Grette	1985	61.8 (unlogged) 33.3 (young 2nd-growth) 31.7 (middle-aged 2nd-growth) 40.5 (old 2nd-growth)	Olympic Peninsula, Washington
Heifetz et al.	1986	73	Alaska
Keller and Tally	1979	50-90	northwest California
Murphy	1983	69 (old-growth) 87 (buffered) 63 (clear-cut)	southeast Alaska
Rainville et al.	1985	80	Idaho Panhandle
Ralph et al.	1991	50 (gradient/confinement=1) 63 (gradient/confinement=2) 52 (gradient/confinement=3) 40 (gradient/confinement=4)	Washington
Tripp and Poulin	1986b	73 (non-debris torrented) 40 (debris torrented)	Queen Charlottes

Table 2a: LWD frequency (pieces/channel width) in unmanaged streams. Blank cells indicate no values could be specified. Criteria used to define LWD vary between studies (Table 4).

CHANNEL WIDTH (m)	1	2	3	4	5	6	7	8
1								1.2[a]
2							1.15	
3				0.46				
4	2.44	1.15						
5	2.38	1.23	3.1				2.47	
6	2.33							
7	2.28		2.59					
8	2.25			2.92		2.42		2.21[b]
9	2.22	3.61						
10	2.19					3.12		
11	2.16		2.2			3.42	7.88	
12	2.14							
13	2.12	4.35		3.20				
14	2.10							
15	2.08				2.53	2.13		
16	2.07							
17	2.05							
18	2.04		7.02	3.95				
19	2.03				6.45			2.09[b]
20	2.01					9.03		
21								
22	1.99							
23								
>23 25.6	1.95	10.88 (26m)			7.33 (24m)	14.38 (31m)		0.69[c]

Note: All values converted to pieces/channel width by multiplying channel width (m) and the number of LWD pieces/m (pieces/m derived from studies 1-8, listed below Table 2b).

Table 2b: Large woody debris frequency (pieces/100m) in unmanaged streams by channel width. Blank cells indicate no values could be specified. Criteria used to define LWD vary between studies (Table 4).

<table>
<tr><th>CHANNEL WIDTH (m)</th><th>1</th><th>2</th><th>3</th><th>4</th><th>5</th><th>6</th><th>7</th><th>8</th></tr>
<tr><td>1</td><td></td><td></td><td></td><td rowspan="5">18.4</td><td></td><td></td><td></td><td>60[a]</td></tr>
<tr><td>2</td><td></td><td></td><td></td><td></td><td></td><td rowspan="2">48</td><td rowspan="13">26[b]</td></tr>
<tr><td>3</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>4</td><td>61.05</td><td>25</td><td></td><td></td><td></td><td rowspan="3">52</td></tr>
<tr><td>5</td><td>47.56</td><td>25</td><td>62</td><td></td><td></td></tr>
<tr><td>6</td><td>38.77</td><td></td><td></td><td rowspan="5">38.9</td><td></td><td></td></tr>
<tr><td>7</td><td>32.62</td><td></td><td>37</td><td></td><td></td><td rowspan="9">73</td></tr>
<tr><td>8</td><td>28.09</td><td></td><td></td><td></td><td>29.5</td></tr>
<tr><td>9</td><td>24.62</td><td>41</td><td></td><td></td><td></td></tr>
<tr><td>10</td><td>21.88</td><td></td><td></td><td></td><td>32.8</td></tr>
<tr><td>11</td><td>19.66</td><td></td><td>20</td><td rowspan="5">25.6</td><td></td><td>31.1</td></tr>
<tr><td>12</td><td>17.84</td><td></td><td></td><td></td><td></td></tr>
<tr><td>13</td><td>16.31</td><td>34</td><td></td><td></td><td></td></tr>
<tr><td>14</td><td>15.01</td><td></td><td></td><td></td><td></td></tr>
<tr><td>15</td><td>13.89</td><td></td><td></td><td>17.3</td><td>14.6</td><td rowspan="9">11[b]</td></tr>
<tr><td>16</td><td>12.92</td><td></td><td></td><td rowspan="5">22.6</td><td></td><td></td><td></td></tr>
<tr><td>17</td><td>12.08</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>18</td><td>11.34</td><td></td><td>39</td><td></td><td></td><td></td></tr>
<tr><td>19</td><td>10.66</td><td></td><td></td><td>34.3</td><td></td><td></td></tr>
<tr><td>20</td><td></td><td></td><td></td><td></td><td>44.5</td><td></td></tr>
<tr><td>21</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
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<tr><td>>23</td><td></td><td>42
(26m)</td><td></td><td></td><td>30.17
(24m)</td><td>45.8
(31m)</td><td></td><td>3[c]</td></tr>
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Table 17: Observed egg burial depths for several salmonid species.

AUTHOR	DATE	SPECIES	EGG DEPTH (cm)	LOCATION
Koski	1966	coho	17.8-27.9	coastal Oregon
Tripp and Poulin	1986a	coho	0-45 x=29	Queen Charlottes
van den Berghe and Gross	1984	coho	8.9-26.7[a] x=15.3	Snohomish County, WA
Tripp and Poulin	1986a	chum	0-45 x=25	Queen Charlottes
McNeil	1962	pinks	7-37.5 x=22.3	southeast Alaska
Crisp and Carling	1989	brown trout	5-12[a]	England
Grost et al.	1991a	brown trout	2-23 x=9-12	southeast Wyoming[b]
Ottaway et al.	1981	brown trout	0-25[a]	England
Reiser and Wesche	1977	brown trout	x=9-12	southeast Wyoming[b]
Crisp and Carling	1989	sea trout	5-25[a]	England and Wales
Ottaway et al.	1981	sea trout	3-22[a]	England
Crisp and Carling	1989	Atlantic salmon	15-25[a]	England and Wales
Ottaway et al.	1981	Atlantic salmon	10-18[a]	England

[a] Range of egg burial depths correlated to female size
[b] The sites examined in these 2 studies were identical

Table 16: Approximate depths of gravel scour in salmonid streams surveys.

AUTHOR	DATE	SCOUR DEPTH (cm)	LOCATION
Lisle	1989	up to 32	north coastal California
Nawa and Frissell	1991	x = 7.35	southwest Oregon
Seattle Water Department	1991	up to at least 22	Cedar River, Washington
Tripp and Poulin	1986a	5[a] 14[b] 21[c] 28[d] 32[e]	Queen Charlotte Islands

[a] streams with no mass wasting and low gradient
[b] streams with no mass wasting and high gradient
[c] streams with intermediate mass wasting and moderate gradient
[d] streams with slides and moderate gradient
[e] streams with debris torrents and high gradient

Table 15: Classification of previously surveyed streams in managed forests as good or poor in terms of substrate composition based on proposed target condition (11% fines <0.85mm).

STUDY	LOCATION	STREAMS WITH GOOD SUBSTRATE COMPOSITION	STREAMS WITH POOR SUBSTRATE COMPOSITION
1	Nooksack River Basin, 1982	6	10
1	Nooksack River Basin, 1983	5	7
2	Yakima River Basin	9	9
3	Clearwater River Basin, January	0	5
3	Clearwater River Basin, June	0	5

Description of studies

1. Schuett-Hames and Schuett-Hames (1984) Surveys of streams draining managed forests in the Nooksack River Basin
2. Watson (1991) Survey of streams draining managed forests in the Upper Yakima River Basin
3. Tagart (1976) Data collected during survival-to-emergence studies in Clearwater River Tributaries

Table 14: Observed values for % fines in unmanaged stream surveys.

AUTHOR	DATE	FINE SIZE (mm)	% FINES	LOCATION
Edgington	1984	<0.833	6.3 6.5 6.9 7.3 9.8 12.3 12.9 13.6	southeast Alaska
Koski	1966	<0.833	22.7 27.7 28.4	coastal Oregon
Sheridan et al.	1984	<0.83	9.65+-5.19[c]	southeast Alaska
Cederholm	1991	<0.85	11.38[a] 14.50[a]	Main and South Fork Hoh River
Cederholm and Reid	1987	<0.85	6.37 SD = 2.61	Olympic National Park
Hatten	1991	<0.85	6.6 11.0 11.0 11.2 14.5	Hoh River Tributaries
Tripp and Poulin	1986a	<0.85	4.3[d], 4.4[d] 4.6[e], 5.6[e]	Queen Charlotte Islands
Adams and Beschta	1980	<1	10.6-29.4[b]	Oregon Coast Range
Koski	1984	<1	7-9	southeast Alaska
Scrivener and Brownlee	1989	<1.19	<8	Carnation Creek
Scrivener and Brownlee	1989	<2.38	<15	Carnation Creek
Ringler and Hall	1988	<3.33	45.2	coastal Oregon
Koski	1984	<4.0	6.7 7.4 16.0 18.4	southeast Alaska

[a] Note that affected areas actually had less fines in this study (11.35 and 11.69), [b] Amount by weight, [c] used wet and dry methods, [d] non-mass wasting, [e] mass wasting upstream

NCASI	1984	<0.85	rainbows	STE of eggs in troughs
Peterson and Metcalfe	1981	0.06-0.5 0.5-2.2	Atlantic Salmon,	STE of eggs in troughs
Phillips et al.	1975	1.0-3.0	coho, steelhead	STE of alevins in troughs
Reiser and White	1988	<0.84 and 0.84-4.6	chinook, steelhead	eyed and green egg survival to alevin stage
Shepard et al.	1984	<6.4	bull trout	STE of eggs in troughs
Sowden and Power	1985	<2.0	rainbows	pre-emergent fry in a groundwater fed streambed
Tagart	1976	<0.85	coho	emergence from natural redds
Tagart	1984	<0.85	coho	emergence from natural redds
Tappel and Bjornn	1983	<0.85 and <9.5	chinook and steelhead	STE of water hardened and eyed eggs
Weaver	in review	<6.35	bull trout	artificial redds in natural streams
Weaver and Fraley	in review	<6.35	cutthroat	artificial redds in natural streams
Weaver and White	1985	<9.5	bull trout	STE in artificial redds and troughs

Table 13: Studies examining salmonid egg and alevin survival with respect to percent fines.

AUTHOR	DATE	FINES CRITERIA (mm)	SPECIES	COMMENTS
Bjornn	1969	<6.35	chinook, steelhead	STE of green eggs and alevins in troughs
Burton et al.	1990	<6.3	chinook, rainbows	STE of eggs in egg baskets placed in natural redds
Cederholm and Salo	1979	<0.85	coho	STE of eggs in troughs
Cederholm et al.	1981	<0.85	coho	STE of eggs in troughs and natural redds
Hall and Campbell	1969	<0.83, 1.0-3.0	steelhead, coho	STE of eggs and fry in troughs and natural redds
Hausle	1973	<2.0	brook trout	STE of eggs in troughs
Hausle and Coble	1976	<2.0	brook trout	STE of alevins in troughs and eggs in natural redds
Irving and Bjornn	1984	<0.85, <6.35 and <9.5	cutthroat, rainbows, kokanee	STE of water-hardened embryos in troughs
Koski	1966	<3.3	coho	emergence from natural redds
Koski	1975	<3.3	chum	STE of eggs in spawning channel
Koski	1981	<3.3	chum	STE of eggs in spawning channel
MacCrimmon and Gots	1986	<4.0	rainbows	STE of eyed eggs in cells with constant flow to eggs
McCuddin	1977	<6.4 and 6.4-12.0	chinook, steelhead	STE of eggs in troughs

Table 12: Substrate size fraction with the highest correlation to salmonid survival-to-emergence in studies comparing the use of several different size fractions to predict survival-to-emergence.

AUTHOR	DATE	SPECIES	TYPE OF STUDY	MOST DAMAGING SIZE FRACTION (mm)
Irving and Bjornn	1984	kokanee, rainbows, cutthroat	lab	<0.85
Koski	1966	coho	field	<3.33
Reiser and White	1988	steelhead and chinook	lab	<0.85
Young et al.	1991a	cutthroat	lab	<6.3 and <3.35

Table 11: General conclusions from studies examining the relative benefits of different substrate composition indices to predict salmonid survival-to-emergence and changes in streambed composition. Note that these studies include original research, reviews of existing data and theoretical discussions (see description of studies below table).

<u>legend:</u>
F - fredle index, D - geometric mean particle size, % fines - the percent of substrate smaller than a particular size, > -index on the left of this symbol is a better predictor than that on the right, = -differences between the two indices could not be determined

AUTHOR	DATE	SURVIVAL-TO-EMERGENCE	CHANGES IN STREAM BED
Beschta	1982		F > D = % fines
Chapman	1988	F > D	
Lotspeich and Everest	1981		F > D
Scrivener	in review		F > D =% fines
Shirazi and Seim	1981	D = % fines	D > % fines
Young et al.	1990	F = D	
Young et al.	1991a	D > F > % fines	% fines > F = D

<u>Description of studies</u>

Beschta (1982) - used data from past studies to comment on Shirazi and Seim (1981)

Chapman (1988) - reanalyzed field and laboratory studies to determine the relative benefits of D and F

Lotspeich and Everest (1981) - used D and F to express composition of laboratory mixtures and related F to data from previous studies.

Scrivener (in review) - comparison of methods using existing Carnation Creek freeze-core data and estimates of STE from fry emmigration and snorkel surveys

Shirazi and Seim (1981) - expressed data from previous studies using D and conducted stream sampling

Young et al. (1990) - reanalyzed Phillips et al.'s (1975) and Tappel and Bjornn's (1983) STE data.

Young et al. (1991a) - 1. STE in artificially constructed egg pockets (centrums) in troughs with 3 replicates of each of 31 different substrate compositions over 2 years.
2. Freezecore, McNeil cylinder, and shovel samples from egg pockets, redds and next to redds from 2 creeks over 3 years

Table 10: Approximate survival-to-emergence of salmonids at 11% and 16% fines <0.85mm estimated by eye from figures and summary data presented in reviews or original research.

AUTHOR	DATE	SPECIES	TYPE OF STUDY	% SURVIVAL AT 11% FINES <0.85mm	% SURVIVAL AT 16% FINES <0.85mm
Irving and Bjornn	1984	cutthroat	lab	0-40[a]	not indicated
Irving and Bjornn	1984	rainbows	lab	10-36[a]	0-18[a]
Reiser and White	1988	chinook-green eggs	lab	10	5
Irving and Bjornn	1984	kokanee	lab	17-70[a]	0-50[a]
Tappel and Bjornn	1983	chinook	lab	22->80[a]	0-72[a]
Cederholm et al.	1981	coho	lab	24	11
Reiser and White	1988	steelhead -green eggs	lab	24	20
Tappel and Bjornn	1983	steelhead	lab	39-71[a]	7-42[a]
Cederholm et al.	1981	coho	field	46	38
Tagart	1976	coho	field	54	34
Reiser and White	1988	steelhead -eyed eggs	lab	60	58
Koski	1966	coho	field	70[b]	54[b]
Hall and Campbell	1969	coho	field	not indicated	69

[a] Range of values for survival to emergence depends on % fines <9.5mm (Tappel and Bjornn 1983), values estimated from isolines predicting survival-to-emergence based on observed values

[b] data presented in Chapman (1988)

Sedell et al.	1984	after Duff and Cooper (1976)
Sullivan	1986	identified 5 pool types based on flow characteristics, area and cause of formation
Tripp and Poulin	1986b	1981, 1982 - areas with greatest depth, low gradient and placid surface 1983, 1984 - after Bisson et al. (1982)

Table 9: Criteria for the identification of pools used in different studies.

AUTHOR	DATE	POOL CRITERIA
Andrus et al.	1988	volume >1m^3, average depth >30cm
Beechie and Sibley	1990b	after Sullivan (1986)
Beechie and Wyman	1992	primary pools - width >50% low-flow channel width, depth >36" (>24" small streams); all pools - after Bisson et al. (1982) and Sullivan (1986)
Bilby and Bisson	1987	after Bisson et al. (1982)
Bilby and Ward	1989	after Bisson et al. (1982)
Bilby and Ward	1991	after Bisson et al. (1982)
Bisson et al.	1982	identified 6 pool types based on mechanisms of formation and morphology
Bisson et al.	1988	after Bisson et al. (1982)
Carlson et al.	1990	relatively still water where stream bottom is concave and both length and width were >.5 summer stream width
Carman et al.	1984	habitat divided into either pools or riffles, pool quality rating after Platts 1979
Fox	1992	primary pools - width >50% low-flow channel width, depth >36"
Grette	1985	smooth-surfaced habitats characterized by low velocities (included glides and runs)
Hogan	1986	'low areas'
Heifetz et al.	1986	after Bisson et al. (1982)
Johnson et al.	1986	after Johnson and Heifetz(1985)
Kaufmann	1987	after Bisson et al. (1982), 'with modifications'
Lisle	1986	residual depth >0
Murphy et al.	1984	after Bisson et al. (1982)
Ralph et al.	1991	after Sullivan (1986)

Table 8c: Pool spacing (number of channel widths between pools) in unmanaged streams. Criteria used to define a pool vary considerably between studies (Table 9).

AUTHOR	DATE	POOL SPACING (CHANNEL WIDTHS)	BANK FULL WIDTH (m)	GRADIENT (%)	LOCATION
Bilby and Ward	1991	1.15[a] 1.02[a] 1.21[a] 1.62[a]	5 10 15 20	1-18	southwest Washington
Carlson et al.	1990	3.05 1.39 0.62 2.05 1.31 2.22 4.35 0.86 2.63 2.02 2.56	4.1 4.0 4.5 6.1 4.5 4.1 2.3 4.0 3.8 3.8 3.0	2.0 2.1 3.5 3.8 4.4 4.5 5.4 5.6 6.5 7.1 7.4	northeast Oregon

Note: Pool spacing = (100/channel width)/(# of pools per 100m)

[a] LWD-associated pool spacing converted from Bilby and Ward's (1991) regression

Table 8b: Pool frequency (number per 100m) in unmanaged streams by gradient and channel width. Criteria used to define a pool vary considerably between studies (Table 9).

AUTHOR	DATE	POOLS /100M	GRADIENT (%)	BANK FULL WIDTH (m)	LOCATION
Bilby and Ward	1991	17.38[a] 9.77[a] 5.49[a] 3.09[a]	1-18	5 10 15 20	southwest Washington
Carlson et al.	1990	8 18 36 8 17 11 10 29 10 13 13	2.0 2.1 3.5 3.8 4.4 4.5 5.4 5.6 6.5 7.1 7.4	4.1 4.0 4.5 6.1 4.5 4.1 2.3 4.0 3.8 3.8 3.0	northeast Oregon
Grette	1985	7.9[b]	0.8-1.5	not indicated	Olympic Peninsula
Sedell and Everest	1990	2.42[c]	not indicated	not indicated	Columbia River Basin

[a] LWD-associated pool frequency determined using the regression:

$\log_{10}$pool frequency = -0.05*channel width + 1.49 from (Bilby and Ward 1991)

[b] number of pools >10m^2

[c] number of pools >3 feet deep and >25 yards^2

Table 8a: Pool percentages (% of stream surface area comprised of pools) in unmanaged streams by channel width and gradient. Criteria used to define a pool vary considerably between studies (Table 9).

AUTHOR	DATE	POOLS (%)	BANK FULL WIDTH (m)	GRADIENT (%)	LOCATION
Bilby and Bisson	1987	58.8[c]	not indicated	not indicated	Deschutes tributary
Carlson et al.	1990	28 39 21 27 29 12 16 37 20 18 12	4.1 4.0 4.5 6.1 4.5 4.1 2.3 4.0 3.8 3.8 3.0	2.0 2.1 3.5 3.8 4.4 4.5 5.4 5.6 6.5 7.1 7.4	northeast Oregon
Grette	1985	81.1	not indicated	0.8-1.5	Olympic Peninsula
Heifetz et al.	1986	56[c]	x = 6.5[a]	0.1-3.0	southeast Alaska
Johnson et al.	1986	49[d]	not indicated[d]	2.5	southeast Alaska
Murphy et al.	1984	39[c] 50[c] 67[c]	8.3 10.3 6.6	1.0[b]	southeast Alaska
Ralph et al.	[e]	51[c]	3-19	1-18	Washington
Tripp and Poulin	1986b	32.7 25.7	not indicated	1.1-6.0 2.7-9.2	Queen Charlottes
Sullivan et al. (after Dinicola 1979)	1987	20 30 65 40	11 18 7 5	3 4 4 5	Deschutes River Basin

[a] from Murphy (1983)
[b] from Koski (1984)
[c] used pool classification system of Bisson et al. (1982) or Sullivan (1986)
[d] this stream is a subset of those described by Heifetz et al. (1986)
[e] in preparation (1991 ambient monitoring data)

Sedell and Everest	1990[e]	pool frequency	Columbia River basin
Sullivan et al.	1987[d]	% pool area	Deschutes River basin
Tripp and Poulin	1986b[d]	% pool area (regressions), pool depth	Queen Charlottes

[a] published in refereed literature
[b] tribal survey
[c] thesis
[d] agency report
[e] unpublished summary data

Table 7: Presentation of pool frequency and character in fisheries literature.

AUTHOR	DATE	POOL CLASSIFICATION	LOCATION
Andrus et al.	1988[a]	total pool volume, pool volume	coastal Oregon
Beechie and Sibley	1990a[b]	% pool area	north Cascades
Beechie and Sibley	1990b[b]	% pool area (regressions)	north Cascades
Beechie and Wyman	1992[b]	% pool area, primary pool frequency	Skagit River system
Bilby and Ward	1989[a]	% surface area of a pool	western Washington
Bilby and Ward	1991[a]	% surface area of a pool, LWD/pool frequency	southwest Washington
Cardoso and Ralph	1991[e]	% pool area, relative pool frequency	Washington
Carlson et al.	1990[a]	pool frequency, % pool area, pool/riffle ratio, total pool volume	northeast Oregon
Carman et al.	1984[d]	% pool area	Capitol Forest, WA
Fox	1992[b]	'primary' pool frequency	Mt. Rainier National Park
Grette	1985[c]	% surface area of a pool, pool frequency	Olympic Peninsula
Hogan	1986[d]	pool spacing, relative pool depth	Queen Charlottes
Kaufmann	1987[c]	residual pool volume	Oregon Coast Range
Lisle	1986[a]	pool frequency, residual pool depth, % of channel length	Prince of Wales Island, Alaska
Ralph et al.	1991[d]	relative frequency, % pool area	Washington

Table 6: Average debris volume index (m^3 per LWD piece) in unmanaged and managed stream surveys. Criteria used to define LWD vary between studies (Table 4).

CHANNEL WIDTH (m)	1 unmanaged	2 unmanaged	3 managed
4	0.25	1.4	0.29
5	0.48	1.5	0.51
6	0.71		0.31
7	0.94		0.32
8	1.17		0.54
9	1.40	1.5	0.33
10	1.63		0.49
11	1.86		0.64
12	2.09		0.71
13	2.32	1.8	
14	2.55		
15	2.78		0.59
16	3.01		0.26
17	3.24		
18	3.47		0.52
19	3.70		0.55
20	3.93		
21			
22	4.39		0.78
>22		2.4 (26m)	

25.6 5.22

Description of studies in Table 6

1. Based on Bilby and Ward's (1989) regression (debris volume index = 0.23*channel width - 0.67) for western Washington
2. Robison and Beschta (1990a) for unmanaged streams in southeast Alaska
3. Beechie (unpublished) for managed streams in the Stillaguamish and Snohomish basins.

Description of studies in Table 5

1. Based on Bilby and Ward's (1991) regression for clear-cut

   ($\log_{10}$LWD frequency = -1.35 channel width + 0.50)

2. Based on Bilby and Ward's (1991) regression for second-growth

   ($\log_{10}$LWD frequency = -1.23 channel width + 0.28)

3. Cederholm et al. (1989)- Olympic Peninsula
4. Cederholm and Peterson (1985)- Olympic Peninsula
5. Beechie (unpublished data)- 1988-1990 Stillaguamish and Snohomish system survey
6. Beechie and Wyman (1992)- averaged at each channel width over 4 Skagit River watersheds

Table 5: Large woody debris frequency (pieces/channel width) in existing stream surveys of managed forests. Criteria used to define LWD vary between studies (Table 4).

CHANNEL WIDTH (m)	1	2	3	4	5	6
4	1.95	1.39			1.20	0.31
5	1.80	1.32			1.80	1.51
6	1.68	1.26			3.20	
7	1.60	1.22			3.38	
8	1.53	1.18		2.13	2.84	2.51
9	1.47	1.15		2.88	3.63	0.31
10	1.41	1.12		3.17	2.63	1.20
11	1.37	1.10			3.24	
12	1.33	1.08	1.99	2.39	4.68	
13	1.29	1.06		4.64		
14	1.26	1.04				
15	1.23	1.02	3.31	3.02	6.23	
16	1.20	1.01			0.78	
17	1.17	0.99	5.51			
18	1.15	0.98	3.46		4.32	
19	1.13	0.97			1.51	
20	1.11	0.96				
21	1.09	0.95				
22	1.07	0.94			12.4	
23	1.06					

Note: All values converted to pieces/channel width by multiplying channel width (m) and the number of LWD pieces/m (pieces/m derived from the studies listed below)

Lestelle and Cederholm	1984	>10	none	Clearwater River headwaters
Lienkaemper and Swanson	1987	>10	>1.5	Oregon Cascades
McMahon	1991	>5->20[a]	>1->2.4[a]	Oregon
Murphy et al.	1986	>10	>1	southeast Alaska
Murphy and Koski	1989	>10	>1	southeast Alaska
Osborn	1981	>10	>0.3	Clearwater River drainage
Potts and Anderson	1990	>10	none	western Montana
Ralph et al.	1991	10-50	>3	Washington
Ralph et al.	[b]	10-50	>3	Washington
Robison and Beschta	1990a	>20	>1.5	southeast Alaska

[a] Criteria differed in different studies included in the data base

[b] 1991 ambient monitoring data (in preparation)

Table 4: Criteria for the measurement of large woody debris used in different studies.

AUTHOR	DATE	DIAMETER (cm)	LENGTH (m)	LOCATION
Andrus et al.	1988	>10	>1	coastal Oregon
Beechie and Sibley	1990b	>20	>1.5	north Cascades
Beechie and Wyman	1992	>10	>2	Skagit River system
Beschta and Veldhuisen	1991	>15	>2	central Oregon Coast Range
Bilby and Ward	1989	>10	>2	western Washington
Bilby and Ward	1991	>10	>2	southwest Washington
Bisson and Sedell	1984	>10	<3 >3	western Washington
Bryant	1982	>10	none	southeast Alaska
Carlson et al.	1990	>10	>1	northeast Oregon
Cederholm et al.	1989	>10	>3	Olympic Peninsula
Cederholm and Peterson	1985	>10	>3	Olympic Peninsula
Fox	1992	>10	>3	Mt. Rainier National Park
Froehlich	1972	>10	>0.3	Oregon Cascades
Grette	1985	>10	>3	western Olympic Peninsula
House and Boehne	1986	>10	none	Tobe Creek, Oregon
House and Crispin	1990	>15	>3	Tillamook County, Oregon
Keller and Swanson	1979	>10	none	western Oregon, Indiana, and N.C.

Table 3b: Large woody debris (pieces/channel width) by channel gradient in unmanaged streams. Criteria used to define LWD vary between studies (Table 4).

GRADIENT (%)	Robison and Beschta (1990a)	Sullivan et al. (1987)[a]	Murphy and Koski (1989)
0.5	10.88		14.38
1.0	4.35		9.03, 2.42, 2.13
1.5	1.23		
2.0	1.15		3.42
2.5	3.61		
3.0		2.2	3.12
3.5			
4.0		2.59, 7.02	
4.5			
5.0		3.1	

[a] Values estimated from plots in Sullivan et al. (1987) after data from Dinicola (1979).

Note: All values converted to pieces/channel width by multiplying channel width (m) and the number of LWD pieces/m

Table 3a: Large woody debris frequency (pieces/100m) by channel gradient in unmanaged streams. Criteria used to define LWD vary between studies (Table 4).

GRADIENT (%)	Robison and Beshta (1990a)	Sullivan et al. (1987)[a]	Murphy and Koski (1989)
0.5	42		45.8
1.0	34		44.5, 29.5, 14.6
1.5	25		
2.0	25		31.1
2.5	41		
3.0		20	32.8
3.5			
4.0		37, 39	
4.5			
5.0		62	

Description of studies in Tables 2a and 2b

1. Bilby and Ward (1989)-values determined from the regression

$$\log_{10}\text{LWD frequency} = -1.12 \log_{10} \text{channel width} + 0.46$$

Pieces/Meter

2. Robison and Beschta (1990a)
3. Estimated from plots in Sullivan et al. (1987) based on data from Dinicola (1979)
4. Ralph et al. (in preparation)-1991 ambient monitoring data
5. Cederholm et al. (1989)
6. Murphy and Koski (1989)
7. Fox (1992)-in the calculation of pieces/channel width for each stream size range, width was considered to be the midpoint of that range
8. Bilby and Wasserman (1989)
   - [a] estimated values
   - [b] data from Bilby (1985)
   - [c] data from Cederholm (pers. comm.)-width considered to be 23m for the calculation of pieces/channel width

$y = a^x$ iff $x = \log_a y$

$$\log_{10}(LWD) = [-1.12 \log_{10}(width) + 0.46]$$

$$LWD = [-1.12 \log_{10}(width) + 0.46]$$

$$LWD = 10$$