Rates and pathways of recovery for sediment supply and woody debris recruitment in northwestern Washington streams, and implications for salmonid habitat restoration

by

Timothy J. Beechie

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Approved by ______________________________
Chairperson of Supervisory Committee

Thomas Sibley
Derek B. Booth
Peter A. Bisson

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Abstract

Rates and pathways of recovery for sediment supply and woody debris recruitment in northwestern Washington streams, and implications for salmonid habitat restoration

by Timothy J. Beechie

Chairperson of the Supervisory Committee
Professor Susan Bolton
College of Forest Resources

This dissertation describes a strategy for restoring watershed processes. The restoration strategy is driven by the recognition that (1) salmonids are adapted to local environmental conditions, and (2) spatial and temporal variations in landscape processes create a dynamic mosaic of habitats in a river network. Therefore, the strategy's goal is to restore and maintain the habitat-forming processes to which salmonids are adapted. Comparisons of current rates to historic rates highlight areas where restoration is necessary. The strategy then allows for prioritizing restoration actions. Prioritization does not alter the types of actions that may be taken, but alters the sequence in which they are implemented.

Analyses of sediment supply and large woody debris (LWD) recruitment in the context of the natural disturbance regime allow prediction of recovery rates for salmonid habitats. Sediment supply is most significantly affected by changes in mass wasting rates. Stands less than 20 years old have mass wasting rates approximately four times that of mature forest areas, and roads have mass wasting rates approximately 40 times that of mature forest areas. The cumulative effect in a watershed is to increase average annual sediment supply under a forest management regime to about twice that of the natural fire regime. The rate of recovery for sediment supply can be described in terms of average annual travel distance of sediment, which averages about 20 times channel width where channel slope is less than 0.03.

LWD recruitment in the study area has been altered by previous logging practices and conversion to agricultural and urban land uses. Compared to projected LWD recruitment under

the natural fire regime, forest management typically reduces recruitment of LWD large enough to form pools by an estimated 35 to 100%. Recovery rates for LWD recruitment are primarily a function of channel size and the tree species colonizing a disturbed riparian area. Modeling of LWD recruitment through time predicts that active management of riparian forests will most effectively increase recovery rates in large streams, and that active management is unlikely to improve LWD recruitment in any channel less than 15 or 20 m wide.

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Acknowledgments

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Chapter 1. Introduction

Interest in restoring depleted or depressed salmon runs in the Pacific Northwest has grown rapidly in the past decade, primarily because of declining harvest levels, loss of numerous stocks, and petitions to declare several stocks threatened or endangered under the Endangered Species Act (Nehlsen et al. 1991, Lichatowich et al. 1995). Virtually all of the losses or declines were at least partly associated with habitat loss or degradation (Bisson et al. 1992, Gregory and Bisson 1997). Scientists now accept that some of the blame for salmon "crisis" goes to traditional fishery and habitat management, which focused on individual species and habitat characteristics rather than on managing whole ecosystems (Frissell et al. 1997). However, emerging management concepts such as *ecosystem management* or *managing for biodiversity* have not been translated into strategies or actions that are useful to local managers or restoration groups. Hence, the fact that we are in a "crisis" compels local groups to act to save salmon runs, but they continue applying short-term treatments that often prove ineffective because they are unaware of alternative approaches and methods (Bisson et al. 1997).

Short-term restoration measures (e.g., placing log structures, constructing spawning riffles, protecting stream banks) have frequently been unsuccessful in the past because they have not addressed causes of habitat degradation (Frissell and Nawa 1992) or changes that have the largest effect on salmon populations (Reeves et al. 1991). Use of these methods typically assumes that habitat conditions are steady-state and disconnected from large-scale physical or biological processes. However, many changes in habitat conditions have causes that can only be seen when habitats are viewed in the context of large-scale watershed processes. Failure to recognize the integrated nature of physical and ecological processes can cause projects to fail or to have high maintenance costs (Frissell and Nawa 1992, Kauffman et al. 1997). Additionally, using approaches that lack a biological context (i.e., understanding of the species or communities historically present), can result in projects that do not address factors limiting production or that help one species but harm others.

Many authors have recently suggested that restoring and managing watersheds or ecosystems is preferable to managing individual species (e.g., Doppelt et al. 1993, Lichatowich et al. 1995, Reeves et al. 1995). This approach may help avoid adding to a long list of failures in single-species management, which is evidenced by the loss or decline of a large number of Pacific salmon stocks (Nehlsen et al. 1991) as well as of terrestrial species such as the northern spotted owl. As the list of species of concern grows, it is becoming clear that managing habitats for individual species will at best be very complicated, and that managing ecosystems may be one way to simplify the management situation. However, we must still account for local fishery management objectives such as escapement and harvest goals when establishing restoration plans (Lichatowich et al. 1995), and in more extreme cases we may be required to account for species that are listed under the Endangered Species Act (Collins et al. 1994). A lack of sensitivity to such locally important issues may result in less successful restoration in the long-term.

Beechie et al. (1996) developed a strategy for restoring habitats for all salmonids in Pacific Northwest watersheds, while at the same time accounting for local fishery management priorities. The strategy has a goal of restoring natural habitat-forming processes in watersheds, which guides the identification of needed restoration actions. Restoration actions may then be prioritized based on a biological indicator of local importance (e.g., a depressed stock). In practice, this restoration framework accounts for local land uses, and recognizes the role of landscape processes at large spatial scales. In addition, it provides a context for evaluating restoration options where land uses constrain the types of restoration that can be undertaken. The approach can lead to restoring all habitat-forming processes where there are few land use constraints, or to restoring only selected habitat-forming processes where land uses impose severe constraints on restoration activities.

This use of the term "restoration" is consistent with the definition proposed by Cairns (1990), who argued that restoration may take one of four general forms: (1) allow to recover naturally, (2) restore the original ecosystem, (3) restore selected components of the ecosystem, or (4) create a new ecosystem. Although only the first two are strictly consistent with the goal of restoring natural processes, the third and fourth forms of restoration may be most common in urban or agricultural areas where restoration options are constrained by land use. Other authors have argued that "restoring to a new ecosystem" and perhaps even "restoring selected components of an ecosystem" do not fit the definition of *restoration* (e.g., Gore 1985, Yount and Niemi, 1990, Jordan 1995). Rather, they suggest that *restoration* must imply returning to some previous condition, and that alternate terms such as *recovery enhancement* (Gore 1985) or *rehabilitation* (Jordan 1995) are more appropriate where systems cannot be restored to "natural" conditions. I have chosen to use Cairns definition of restoration (including all of its four forms) throughout this dissertation.

This dissertation has three main objectives. First, it summarizes a restoration approach that is focused on *causes of degradation* rather than on *effects of degradation* (Beechie et al. 1996), and further explains the physical and biological basis for restoring watershed processes (Chapter 2). It also describes the steps necessary to apply the approach, and describes how this approach may be viewed as a compliment to other diagnostic tools. The second objective is to develop an understanding of anthropogenic disturbance to salmonid habitats in the context of a natural disturbance regime for two habitat-forming processes: sediment supply and LWD recruitment (Chapters 3, 4 and 5). Chapter 3 defines disturbance and recovery of habitat-forming processes. Chapters 4 and 5 describe typical mechanisms and magnitudes of disturbance for sediment supply or LWD recruitment, and compare rates of disturbance under forest management and natural fire regimes. Each chapter also presents a theoretical and empirical basis for describing rates and pathways of recovery, which can then be used to estimate rates of salmonid habitat recovery in streams. The final objective of this dissertation is to use descriptions of recovery rates and pathways as an aid in planning the restoration of salmonid habitat. Chapter 6 describes two example applications of these recovery pathways in the North Cascades of Washington state.

Chapter 2. An approach to restoring salmonid habitat-forming processes in Pacific Northwest watersheds

This chapter presents a restoration approach that is focused on diagnosing and treating *causes of habitat degradation* rather than *effects of habitat degradation*. The approach presented here does not deviate significantly from the original approach described in Beechie et al. (1996). However, this chapter further explains the physical and biological basis for restoring watershed processes based on historical reconstruction of habitat-forming processes. It does not claim that all of the scientific tools are available to fully reconstruct historical rates and magnitudes of habitat-forming processes, but it recognizes that identification of changes in processes is essential if restoration is to be sustainable and cost-effective. This chapter also describes the steps for historical reconstruction of habitat-forming processes, and for identifying and prioritizing restoration tasks. Finally, it compares the approach of diagnosing changes in habitat-forming processes to two other diagnostic approaches that focus on effects of degradation, and it discusses its utility as either an alternative or a compliment to them.

Some implications of salmonid adaptations to a dynamic environment

Beechie et al. (1996) contend that watershed or aquatic habitat restoration should be based on two underlying physical and biological principles:

1. Spatial and temporal variations in landscape processes create a dynamic mosaic of habitat conditions in a river network (Naiman et al. 1992a, Benda 1994, Reeves et al. 1995).
2. Salmonid stocks are adapted to local environmental conditions (Miller and Brannon 1982, Healey 1991).

Together, these statements imply that salmonid stocks are adapted to spatially and temporally variable habitats, and may further imply that such variability is important to their long-term survival (Reeves et al. 1995). This suggests that restoration should allow for spatial and temporal variability in habitat conditions, but also that the variability should be appropriate to the natural potential of an area. This view differs from traditional habitat management approaches that use such methods as statewide habitat standards or constructed habitats.

Development of habitat standards is common in regulatory processes, resulting in collections of standards that managers rely on as indicators of habitat degradation (Bisson et al. 1997). While such standards may be based on some set of biological preferences (e.g., preferred temperature ranges or pool area), they rarely account for different natural potentials among watersheds or stream reaches. That is, they do not account for the fact that many locations in a channel network are naturally incapable of producing preferred or optimum habitat conditions. Moreover, they do not recognize that salmonids are adapted to their local environments, which influences local salmonid habitat preferences and to some extent determines the genetic and behavioral diversity of salmonids in the region. Empirical data show that stocks of the same salmonid species are adapted to local differences in habitat characteristics such as temperature regime or flow regime (Wood 1995, Healey and Prince 1995), suggesting that managing for a single standard may be inappropriate for many stocks within a region.

Traditional approaches to salmonid habitat restoration or enhancement focus on repairing or augmenting specific habitat conditions rather than on restoring landscape processes that form and sustain salmonid habitat. By focusing on conditions, these approaches typically lead to engineering solutions designed to create or modify habitats so that they do not move in space or change over time (e.g., protecting stream banks or installing woody debris structures). However, many natural habitats such as off-channel ponds or debris-formed pools that appear relatively stable at one point in time have been created by dynamic processes. Moreover, processes such as channel migration or LWD recruitment continually recreate these habitats over time (e.g., Peterson and Reid 1984, Benda 1994, Abbe and Montgomery 1996). Therefore, attempts to build "stable" habitats may interrupt processes that maintain a diversity of habitats over time.

Processes underlying the dynamic nature of Pacific Northwest aquatic environments have recently received more rigorous consideration, both with respect to the geomorphological patterns they create within channel networks (Benda and Dunne 1997a, b) and to their implications for management of aquatic resources (Reeves et al. 1995, Beechie et al. 1996). For example, simulations of sediment supply and routing in channel networks illustrate the linkages between episodic sediment supply in the headwaters and delayed aggradation in lower river reaches (Benda and Dunne 1997a, b). Traditional restoration approaches focusing on in-channel habitat characteristics may miss such linkages, and therefore fail to recognize the root cause of degradation (Frissell and Nawa 1992, Reeves et al. 1995). This frequently causes project failure or high maintenance costs. This implies that sustainable habitat restoration should focus on habitat-forming processes rather than on engineered solutions, and that management of large scale features that influence the rates and magnitudes of habitat-forming processes is a more logical approach to salmonid habitat management (Reeves et al. 1995).

The need to focus on habitat-forming processes

Despite our understanding that changes in habitat typically cause changes in biota and that habitat degradation is at least partially responsible for declines in salmon abundance (Bisson et al. 1992), there is little in the way of a theoretical basis for describing rates and pathways of recovery for either the biota or the environment that supports them (Cairns 1990). As a consequence, predicting the biological outcome of actions designed to restore habitat-forming processes is difficult because we cannot yet predict the physical outcome of those actions. From a natural resource management perspective, a better understanding of the processes by which habitats are altered and by which they recover is important for (1) better understanding the likely geomorphological outcome of watershed or stream restoration work and (2) more effectively targeting areas where restoration can provide significant biological benefits.

An understanding of habitat-forming processes is also missing from several other approaches to diagnosing changes in habitat conditions in the Pacific Northwest. Lichatowich et al. (1995) presented an approach to diagnosing and treating depleted salmonid populations that focused primarily on historical reconstruction of salmonid life history patterns and habitat conditions. They suggested that a description of historical habitat conditions be included in any restoration plan, although such descriptions could be qualitative because of the lack of historical data. Other methods assess changes in biotic communities or habitat conditions relative to reference site conditions (e.g., Karr 1991), which also focus on conditions in stream channels as diagnostics of habitat degradation. While these approaches help identify the types of degradation that occur at a site, they do not directly identify sources of problems. Thus, it remains to identify what types of land use are causing habitat degradation, where the land uses are located, and what types of restoration actions may be required to solve the problem.

Other approaches to restoring and managing salmonid habitats begin with broad guidelines for land use, but address neither stream habitat conditions nor the habitat-forming processes (e.g., Reeves et al. 1995, Gregory and Bisson 1997). In general, these approaches stress the importance of establishing refugia and mimicking the natural disturbance regime as closely as possible. These approaches are valuable for redirecting land management practices of large land holders such as the USDA Forest Service, but they provide little guidance for restoration planning in watersheds with diverse ownerships. As with in-stream diagnostics, they do not identify the causes of habitat degradation and cannot help identify restoration tasks.

These two general types of management tools (i.e., in-stream diagnostics and revised land use patterns) are important elements of restoration planning, but neither identifies specific actions needed to restore and maintain salmonid habitats. Where land uses cannot be adjusted to mimic natural disturbance patterns, restoration planning will also require identification of specific causes of degradation in order to locate and design appropriate restoration actions. Thus, there is a need for analyses that focus on causes of habitat degradation (i.e., habitat-forming processes), which fills an information gap between diagnostics of habitat condition and broadly defined land management plans.

An approach to diagnosing and restoring disrupted habitat-forming processes

A process-oriented restoration goal

In light of the preceding arguments, Beechie et al. (1996) proposed a *restoration goal* of restoring and maintaining the landscape processes that formed and sustained the habitats to which salmonid stocks are adapted, including such processes as sediment supply, recruitment of large woody debris from the riparian forest, channel migration and avulsion, and other processes that influence stream habitat characteristics. This approach is a departure from traditional restoration and enhancement methods that attempt to create specific habitat attributes that are static in space and time (e.g., engineered LWD structures to create pools, rip-rapped banks to resist erosion). Instead, this approach is focused on the natural potential of a stream channel, as well as on how land use affects channel characteristics (Sedell and Luchessa 1982, Hasfurther 1985, Harvey and Watson 1986, Newbury and Gaboury 1988, Beechie et al. 1994).

The intent of this goal is to help avoid errors such as those made in the past when we tried to create habitat conditions that were "good for salmon." For example, widespread large woody debris (LWD) removal during the 1970s and early 1980s was intended to facilitate upstream migration of adult salmon, but the practice also affected juvenile rearing habitats by reducing pool abundance and cover complexity. Such mistakes are less likely to occur if the restoration goal is to re-establish processes to which salmonids were adapted. Second, the process-oriented goal should help identify actions that restore habitat for all salmonids (Peterson et al. 1992). This is particularly advantageous where we want to restore a diversity of aquatic species rather than a specific salmonid stock. It also allows management agencies to move away from a limited range of habitat manipulations mostly relevant to one species, thereby avoiding attempts give all stream reaches the same characteristics despite variability in geomorphological "potential" from reach to reach.

The goal also pushes scientists and managers to achieve greater understanding of processes that create and maintain stream habitats, which should also help avoid restoration efforts that attempt to constrain processes that create productive salmonid habitats. For example, attempts to restrict bank erosion at one location along a river often push erosion to another bank, and total sediment supply to the reach may not be reduced. At the same time, such efforts can disrupt the process of channel migration, which creates oxbow lakes or terrace tributary channels (Peterson and Reid 1984). These off-channel habitats provide important overwintering areas that persist for decades (Peterson and Reid 1984, Scarlett and Cederholm 1984). Where channel migration is successfully controlled, new off-channel habitats will not form and existing habitats will eventually develop into forests that do not provide habitat for salmonids. The restoration goal should help avoid restoration projects that disrupt habitat-forming processes, and future restoration efforts should require less maintenance.

Locally defined restoration priorities

Restoration priorities may appropriately be based on a narrower range of biological objectives, but only to the extent that prioritization remains subordinate to the goal of restoring natural processes (Collins et al. 1994, Lichatowich et al. 1995, Beechie et al. 1996). This is especially important where management goals have economic importance (e.g., for fisheries, the performance of a wild stock that influences regionwide harvest rates), or legal importance to a variety of landowners and commercial ventures (e.g., the potential listing of a species under the Endangered Species Act). Such issues can have tremendous impacts on local economies and warrant consideration when restoration plans are developed.

Prioritization does not alter the types of restoration identified under the goal of restoring natural processes. Rather it alters the sequence in which restoration actions are carried out (Figure 2-1). For example, Beechie et al. (1994) found that most of the coho salmon (*Oncorhynchus kisutch*) habitat losses in the Skagit River basin were due to elimination of side-

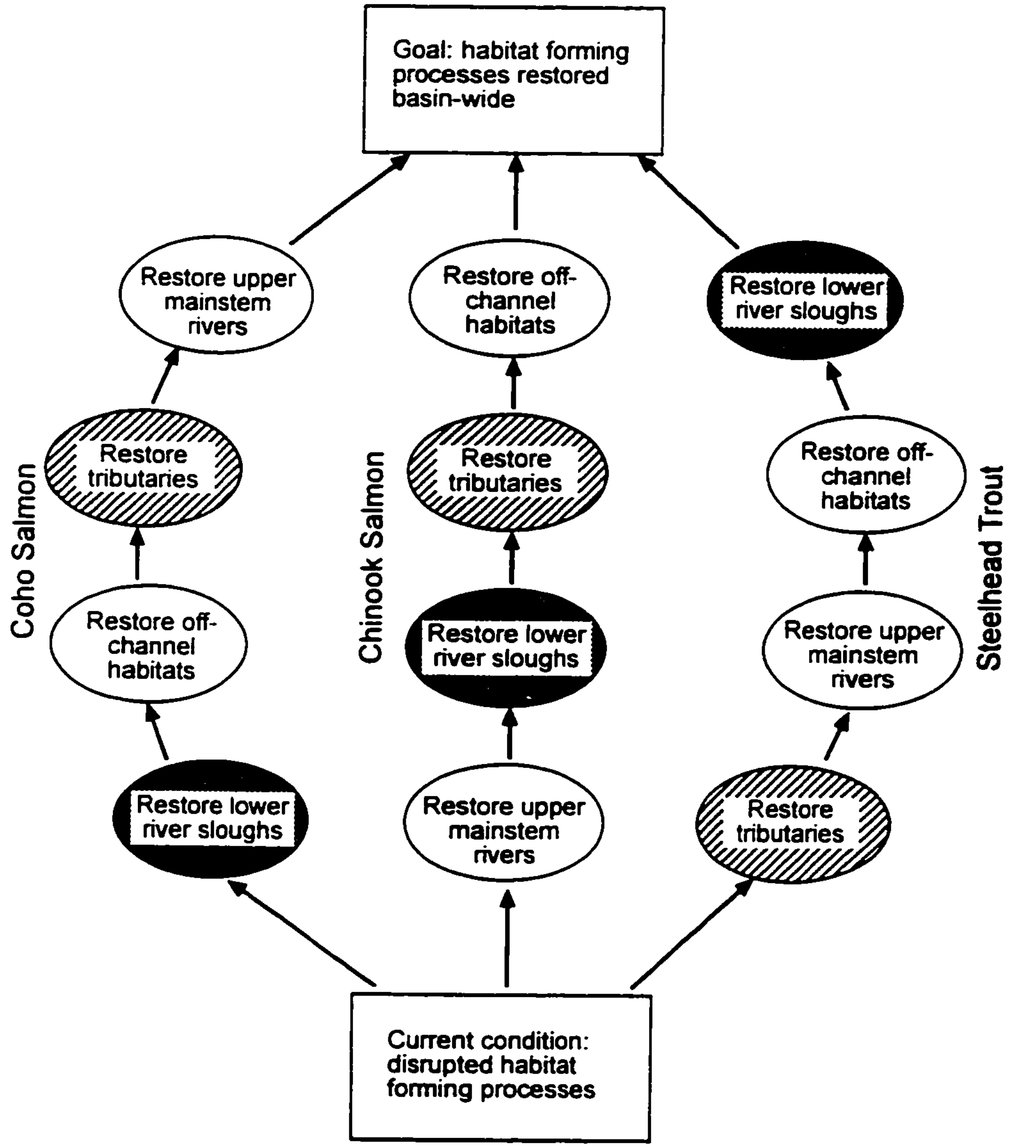


Figure 2-1. The restoration goal guides the identification of actions required to restore and sustain salmonid habitat-forming processes in a watershed, but restoration actions may be sequenced differently depending on local management objectives. This schematic illustrates spatial prioritization of restoration actions, each of which is required to restore all of the habitat-forming processes in a large watershed. Where coho salmon is a priority (pathway on left), restoration efforts may first focus on reconnecting lower river sloughs which once provided significant overwintering habitat in the study area (Beechie et al. 1994). Where steelhead trout are a priority (pathway on right), restoration may focus on tributaries first (Collins et al. 1994). Restoration designed to favor chinook salmon might follow a third sequence of actions (center pathway).

channel sloughs, and that much smaller losses were caused by impassable culverts that isolated tributary habitats. For both situations the first step is to reconnect habitats so migrating fishes have access to historically productive areas. If managers selected coho salmon as the priority species for restoration in this basin, restoration could first focus on reconnecting lower river off-channel habitats, which will recoup the greatest amount of coho production. However, steelhead trout (*O. mykiss*) do not make significant use of slough habitats (Cederholm and Scarlett 1984), and selection of steelhead trout as the priority species would shift the initial focus toward reconnecting tributary habitats. In neither case would a restoration action supersede the goal of restoring natural processes, and completion of the highest priority tasks does not mean that remaining areas or processes can be ignored. However, each sequence would favor recovery of the priority species while also providing benefits for other species.

Overview of the strategy

This strategy focuses analyses on *causes* of habitat degradation rather than on habitats or biota. Although in-stream diagnostics can provide valuable insight into causes of degradation, they cannot by themselves identify where watershed processes are disrupted and what actions may be required to restore them. The approach described here requires analysis of habitat-forming processes at the scale of watersheds in order to identify which processes are disrupted, as well as locations and timing of land use effects on the those processes. Restoration actions can then be identified directly from the results of the analysis. Thus, this approach compliments in-stream diagnostics that assess either habitat characteristics or biotic responses to habitat change.

This strategy is essentially a two-tiered approach, first identifying restoration actions through diagnosis of altered habitat-forming processes, and then prioritizing restoration actions. It was designed to accomplish several objectives. First, it was intended to restore habitat for all salmonid species, while at the same time allowing local managers to sequence restoration activities in a way that favors the recovery of a selected species. Second, it was expected to help avoid failures associated with attempting to engineer habitats that are static in space and time. And finally, the approach was designed to allow managers to evaluate the cost effectiveness of different restoration options. Under this strategy, all restoration actions are identified by assessing changes in habitat-forming processes with respect to their natural rates and magnitudes, and prioritization only alters the sequence of restoration activities. As long as all restoration actions are consistent with the over-riding goal of restoring watershed processes, aquatic habitats will be restored for many species, but in a sequence that favors the local management objective.

Restoration actions that substitute constructed habitats for recovery of habitat-forming processes are inconsistent with the restoration goal. However, in some cases such actions may be used as "emergency measures" if the local biological objective requires it (e.g., for a stock at risk of extinction). These actions are less desirable in restoration because they address symptoms rather than causes (Frissell and Nawa 1992), and are typically expensive relative to the benefit provided. Managers who use constructed habitats in restoration should consider costs, expected benefits, and risk of failure (Beechie et al. 1996). For example, it may be inappropriate to pay for high benefit projects with low probability of success, rather than first preserving a remnant stock with projects that have lower potential benefit but higher probability of success. In any case, careful consideration of the historical context should help in the design of constructed habitats that approximate the conditions to which local salmonid stocks are adapted.

Analysis steps

In practice, application of the strategy includes five steps:

(1) estimate natural rates of habitat-forming processes,

(2) assess changes in rates of habitat-forming processes due to land use,

(3) identify actions required to restore habitat-forming processes,

(4) evaluate probable improvement in local biological indicator (for each task), and

(5) prioritize actions based on costs and potential improvement in biological indicator.

The following sections state the objectives of each step, and briefly describe examples of methods and data that may be used. A summary of the steps and examples of methods are also listed in Table 2-1.

1. Estimate historical rates of habitat-forming processes

Application of this approach first requires an understanding of the behavior of habitat-forming processes under natural conditions. The level of resolution required in the analysis may vary depending on the desired level of sophistication or detail that managers expect as proposed restoration actions. Greater resolution in the analysis should allow managers more flexibility in finding solutions that restore habitat-forming processes while maintaining other land

Table 2-1. Summary of analysis steps and examples of methods or data that may be used in each step.

Step	Examples of methods or data
1 and 2. Historical reconstruction: identify natural processes and land use effects on processes	• Sediment budget (magnitude of sediment supply by process and land use) • Analyses of stand types from aerial photos and maps (e.g., patchy or uniform, maturity, species composition, fire regime, land use alterations) • Temperature models (changes in shade values and stream temperatures over time) • Hydrographs (flood magnitudes, runoff processes) • Analyses of channel and habitat characteristics (changes in channel widths, changes in pool depths or abundance, temporal and spatial agreement between channel responses and floods or sediment inputs)
3. Identify restoration tasks	<u>Passive restoration:</u> • Mapping of mass wasting hazard areas (allow natural recovery of mass wasting rates by avoiding or modifying land uses in hazard areas) • Mapping of recovering riparian areas (allow natural recovery of riparian functions by avoiding or modifying land use) <u>Active restoration:</u> • Inventory blockages to fish passage (restore connectivity of habitats) • Inventory road failure hazards to repair (prevent of future road-related failures). • Map riparian areas to thin or replant (enhance recovery of riparian functions)
4. Evaluate probable effectiveness of restoration tasks with respect to locally chosen biological priorities	• Estimate changes in fish production based on changes in habitat conditions (habitat-based fish production models) • Qualitative estimate of effect of changing temperature regimes (estimate effects of change in temperature on biological indicator species) • Qualitative estimates of fine sediment effects on macro-invertebrate production or survival to emergence
5. Prioritize restoration tasks	• Rank relative effectiveness of different restoration options (i.e., based on greatest change in biological indicator per dollar cost) • Rank options based on shortest recovery time

management objectives. This step uses many existing methods for characterizing fire regimes (Booth 1991), sediment supply rates (Reid et al. 1981, Roberts and Church 1986, Paulson 1997), dynamics of riparian forests (Agee 1988, Featherston et al. 1995), stream temperature regimes (Collins et al. 1994), or rates of other watershed processes. Such methods typically rely on measurement of features such as landslides, forest patches, or stream widths at one or more points in time from aerial photographs, which are then used to estimate rates of various natural processes such as sediment supply (Paulson 1997), fire return (Booth 1991), or channel migration (Collins et al. 1994). Additionally, regional or local rates of natural processes are sometimes already known, but are not used to advantage because analyses tend to focus on conditions rather than processes. For the North Cascades of Washington state, some examples of useful data are unpublished fire history maps or other regional fire data (USDA Forest Service 1996a), sediment budgets or landslide inventories (e.g., Parks 1992, Paulson 1997), and published US Geological Survey hydrologic data (e.g., Williams et al. 1985).

2. Estimate current rates of habitat-forming processes

The second step is to assess how land uses have altered habitat-forming processes. This step identifies mechanisms by which land uses have altered habitat-forming processes, and indicates which of the altered processes are most likely responsible for observed changes in habitat conditions. Methods are typically the same as those used in step 1 in order to facilitate comparisons between historical and present rates of habitat-forming processes (Table 2-1). Comparisons between historic and present rates indicate which habitat-forming processes have been significantly altered from their natural rates, and the largest differences usually indicate which processes have been most disrupted. In general, the level of resolution here should correspond to that of Step 1. However, in some cases greater resolution may be desirable if restoration actions are to be management intensive. For example, active management of riparian forests requires greater understanding of stand dynamics and silvicultural methods than does historical reconstruction of riparian functions.

3. Identify restoration tasks

This step identifies the types of restoration actions (both passive and active) necessary to restore natural habitat-forming processes. The restoration plan should describe the physical and biological objectives of each aspect of restoration, both by location in the watershed and by type of restoration activity (Collins et al. 1994). Each element of the restoration plan should also include detailed descriptions of land areas affected (e.g., how to identify a mass wasting hazard in the field), as well as descriptions of restoration methods (e.g., a range of harvest, thinning, and planting options for restoring riparian vegetation patterns) (Collins et al. 1994). In the absence of significant land use constraints, restoration plans may attempt to fully restore most habitat-forming processes. However, local land-uses may sometimes constrain the degree to which habitat-forming processes can be restored (e.g., urban and agricultural land uses may limit the potential to restore river migration across a floodplain). In these cases restoration plans may target only selected watershed processes in order to avoid dramatic changes in land use and ownership patterns. Where only selected processes will be restored, analysts should identify restoration actions that restore as many processes as possible while accommodating local land use priorities. Such solutions are expected to move toward the restoration goal, but with an understanding that the goal will not be fully achieved and that partial restoration of processes may involve higher costs over the long term.

4. Estimate effectiveness of restoration tasks with respect to local biological objectives

Step four is to evaluate the relative effectiveness of different restoration options in terms of the response of a biological indicator, which is here taken to be a salmonid species of interest. This step is intended to evaluate which of the disrupted habitat-forming processes have had the greatest effect on the biological indicator, and to estimate the degree to which specific types of restoration actions will most effectively restore the freshwater habitats of the species of interest. Effectiveness can in some cases be estimated using habitat-based production models, but in many cases will be evaluated qualitatively based on changes in other factors that affect survival rates (e.g., stream temperature) (Collins et al. 1994). Specific methods required for this step are dependent on the types of processes found to be most severely altered, and on the types of restoration actions considered at a location (Table 2-1).

5. Prioritize restoration tasks

Lastly, the restoration actions described in step three may be prioritized based on the relative benefits to local biological priorities and estimates of cost for different options. Restoration actions may be prioritized in a variety of ways, although the original intent of the strategy was to utilize a simple cost-effectiveness approach (e.g., Beechie et al. 1996). In that approach, projects were ranked in order of decreasing benefit per dollar cost of restoration, where benefits were expressed as the magnitude of the expected increase in a biological indicator. However, other prioritization methods may also be used depending on local preferences. Other factors that may influence priorities are identification of refugia, land-owner willingness to participate, and availability of funds for specific types of projects.

Examples

Sediment supply

The sediment budget is one tool for estimating both historic sediment supply (step 1) and the change in rate of sediment supply due to land use (step 2). Paulson (1997) used sediment budgets to estimate natural rates of various sediment supply processes in several sub-watersheds of the Skagit River basin in Washington state, and to estimate how average annual sediment supply differs between the pre-European settlement fire regime and the past five to six decades of timber harvest practices in the Skagit River basin. For the natural regime, Paulson (1997) found that mass wasting was the dominant sediment supply process in most basins, that much of the spatial variability in mass wasting rates was a function of geology and landform. Mass wasting in the Skagit River basin varied by a factor of three as a function of underlying geology, and over 75% of all mass wasting originated from two easily identified land forms covering less than 25% of the watershed area.

Forest management activities increased average annual sediment supply by 10% to 140% depending on geology and intensity of land use (step 2). In sub-basins dominated by valley filling glacial deposits on the order of 10^1 to 10^2 m thick, recent sediment supply (i.e., over the past 50 years) was dominated by mass wasting associated with clearcut logging. Road locations in those basins were typically on terrace surfaces where mass wasting rates were extremely low, and timber harvest in deeply incised stream valleys has the greatest effect on mass wasting rates. In sub-basins having steep slopes underlain by high grade metamorphic rocks, recent sediment supply was dominated by road-related mass wasting, especially where roads were located such that failures directly entered streams.

Two types of restoration actions were identified from this analysis: active restoration of road-related mass wasting, and passive restoration of mass wasting from deeply incised stream valleys (step 3). Locations of all hazard areas were mapped during the analysis so that restoration actions can be identified with little additional field work. Road-related mass wasting was predominantly caused by failure of sidecast material on slopes >30° or by failure of stream crossing fills. Active restoration should include (1) removal of sidecast material where it has the potential to impact streams, and (2) reconstruction of stream crossings so that culvert failures do not initiate mass wasting. Clearcut-related mass wasting originates primarily from deeply incised stream valleys. Passive restoration on this landform should include avoidance of logging in the most unstable areas and reduced timber removal where the hazard is less (e.g., Collins et al. 1994). The relative importance of active or passive restoration techniques varies by watershed and is a function of the primary cause of mass wasting. For example, where most mass wasting originates from unstable landforms without roads, active restoration of road systems is of lesser importance.

The potential benefit of restored sediment supplies can be estimated by combining in-channel diagnostics with sediment budget information (step 4). For example, Collins et al. (1994) linked changes in mass wasting to changes in steelhead trout parr production using a habitat-based steelhead production model. They estimated that increased sediment supply in the 1980s had filled pools and reduced mainstem rearing habitat capacity by an estimated 35%. From this they inferred that reduced sediment supply would recoup at least the 35% loss in steelhead parr capacity. Among the potential restoration actions identified for sediment supply processes, Collins et al. (1994) concluded that an effort to stop a single large slide was not warranted based on the fact that sediment supply was already declining and the potential remedies were very expensive (step 5). However, reconstruction of 56 road sites and reduced timber harvest in five mass wasting hazard areas (approximately 17% of the watershed area) were considered high priorities for restoration.

Stream temperature

Collins et al. (1994) assessed pre-logging riparian forests using the historical aerial photo record as part of the historical reconstruction of summer temperature regime in the 173 km^2 Deer Creek basin in northwest Washington (step 1). Prior to logging, mature conifer forests covered virtually the entire floodplain along a 13 km stretch of low-gradient river, indicating that virtually no river migration occurred for at least the preceding several decades. Reaches confined by valley walls were also bordered by mature conifer forests. Based on typical heights of mature conifer forests and pre-logging channel widths, Collins et al. (1994) used two empirical temperature models (from Sullivan et al. 1990) to estimate historic maximum temperature and diurnal temperature range in sample reaches throughout the basin. They showed that temperatures at some locations in the basin were historically high due primarily to naturally wide channels where even mature forests provide little shade. Furthermore, their estimates suggested that seven out of eight study reaches historically exceeded the "optimum" salmonid rearing temperature range of 10°C to 14°C (e.g., Bjornn and Reiser 1991), and that at least one reach exceeded the state water quality standard for maximum temperature (18°C) even with a mature conifer riparian forest.

To estimate changes in stream temperature due to land use, Collins et al. (1994) estimated pre-logging (1943) and post-logging (1991) summer stream temperatures and identified which watershed processes were most likely responsible for the changes (step 2). Between 1942 and 1991, estimated maximum stream temperature increased by more than 4°C in 63% of the study reaches as a result of reduced shading, and by less than 3°C in the remaining reaches. Comparisons of timing of floods, sediment inputs from mass wasting events, and riparian logging suggested that riparian logging and flooding were most likely responsible for reduced shade. However, the assessment of impacts of timber harvest on flooding suggested that land use was a minor factor, and Collins et al. (1994) concluded that restoration should focus on restoring the large conifer riparian forests in order to maintain a narrower channel and increased shade levels.

Proposed riparian restoration actions varied by landform and stand type, with several management options for most situations (step 3). Collins et al. (1994) identified 12 combinations of landform and stand type where restoration was warranted, and 21 generalized management prescriptions. They proposed the most intensive management for devegetated floodplains (intensive planting where probability of success is high) and deciduous stands (thinning and interplanting shade-tolerant conifer). Thinning of conifer stands was considered appropriate for young stands where increased growth rates could significantly improve the recovery of stable LWD, but not for older stands where shade and LWD recruitment are presently adequate.

Collins et al. (1994) concluded that riparian restoration actions in reaches with little temperature change between 1942 and 1991 would afford little increase in salmonid survival, whereas restoration in reaches where temperature increased by more than 4°C would provide a relatively large increase in salmonid survival (step 4). Therefore, they suggested that restoration efforts target those areas with maximum temperature increase greater than 4°C (step 5), which encompassed approximately 15 km of stream. They also estimated costs of different restoration options and the length of time to reach a target stand type as a way of comparing the relative cost-effectiveness of restoration options.

Prioritization using the refugia concept

Prioritization of restoration actions may consider factors besides simple benefit or cost-effectiveness estimates, such as the strategic importance of refuge areas. Collins et al. (1994) suggested that recovery of a depressed summer-run steelhead stock would be most efficient if portions of a large watershed were secured as refugia before proceeding with restoration across the entire basin. In that case, refugia were identified as sub-basins where (i) habitats for all freshwater life history stages of steelhead trout were present, (ii) recovery was independent of processes in the rest of the basin, and (iii) restoration was likely to be relatively efficient and rapid. Restoration in these areas had the short-term objective of stabilizing the steelhead trout population, and the long-term objective of providing colonists to the rest of the basin. Upon securing the refugia, the plan then called for restoration of key habitat areas, which were areas that contained a large proportion of the rearing capacity for steelhead trout and coho salmon. These areas had a short-term objective of stabilizing coho salmon and resident trout populations by restoring off-channel rearing areas and small tributaries. The long-term objective was to restore the productive capacity of mainstem and off-channel rearing habitats.

Discussion and Conclusions

There are many different approaches to identifying types of habitat degradation and prioritizing salmonid habitat restoration and management activities. Some approaches focus on effects of degradation as the primary diagnostics (e.g., Lichatowich et al. 1995, Karr 1993), whereas others focus on the change in disturbance patterns between the present land use regime and the historic fire regime (e.g., Reeves et al. 1995). In-stream diagnostics focus on changes in habitat characteristics, and may also indicate which types of habitat-forming processes have been altered. Approaches that directly address land-use patterns target the management of human effects on habitat-forming processes without attempting to understand specific processes or habitat characteristics they create. Neither approach identifies the magnitude of changes to habitat-forming processes or locates specific sites where land uses have altered processes. The process-based restoration strategy presented here focuses on understanding changes to habitat-forming processes, and identifies locations where specific restoration actions are needed to restore habitat-forming processes. Thus, it fills an information gap between in-stream diagnostics of habitat degradation and large-scale assessments of disturbance patterns on a landscape.

Two examples of in-stream diagnostic approaches are patient-template analysis (Lichatowich et al. 1995) and the index of biotic integrity (IBI, Karr 1991). Both focus on diagnosing how habitat *conditions* have changed since European settlement, whereas the restoration of habitat-forming processes focuses on the *causes* of altered conditions. If one accepts the premise that restoring processes (as opposed to characteristics) is critical to successfully restoring habitat conditions over the long term, then some analysis of habitat-forming processes is necessary even when either of the other two diagnostics is used. That is, in order to correct the root causes of degradation we must identify which processes have been disrupted, which land uses have caused the disruption, and where those land uses have caused the disruption. Historical reconstruction of habitat-forming processes provides a benchmark rate for each process, and comparisons of current process rates to historic rates describe the magnitude of changes to those processes. The historical reconstruction also identifies locations of land uses that disrupt habitat-forming processes, which correspond to locations of potential restoration actions.

Many diagnostics of habitat condition do not account for natural spatial or temporal variation in habitat characteristics, although one can describe variability by using a distribution of reach-level habitat conditions (e.g., Lichatowich et al. 1995, Bisson et al. 1997). However, knowing the distribution of current conditions does not provide guidance on how to manage land uses in a way that restores the natural distribution of habitat conditions. The process-based approach to restoration avoids this problem by focusing on how land uses affect habitat-forming processes. The historical analysis identifies how and where land uses have altered habitat-forming processes, which then provides guidance on how to manage land-use effects on natural processes.

The goal of restoring natural habitat-forming processes guides managers away from attempts to control processes, and toward allowing the natural variation in processes to produce variability in habitat characteristics. In addition, restoring habitat-forming processes should restore and sustain conditions over the long term because habitat-forming processes will maintain habitat conditions without continual management intervention.

These comparisons are not intended to suggest that a focus on habitat-forming processes can always substitute for the other two approaches. No single approach to restoration is "best" in all respects, and each approach has advantages in specific situations. IBI has advantages in that a wide array of degraded conditions can be detected with a single sampling method, and it directly measures aspects of biological resources about which managers may be concerned. Patient-template analysis also has a greater focus on biological resources than does the process-based approach, although it is much narrower in scope than IBI. The process-based approach is largely focused on landscape-level processes for identifying degraded conditions, and only incorporates a biological element in prioritizing restoration actions.

A combined approach that employs historical reconstruction of habitat-forming processes, biological diagnostics such as IBI, and an assessment of habitat characteristics and life history patterns of salmonids would increase our understanding of linkages between actions on the landscape and consequences to stream habitats and biota. However, application of a combined approach over large areas would be prohibitively expensive. Therefore, each approach may be used where it provides the greatest advantage. For example, there are many potential causes of habitat degradation in agricultural and urban areas. In those areas IBI could help focus the assessment of habitat-forming processes on the significant causes of degradation in a given watershed, and the historical assessment of habitat-forming processes could be streamlined by giving less attention to those processes which are not significant. The combination of the two approaches would reduce the cost of assessments and would improve the understanding of causes of habitat loss over the use of IBI alone. By contrast, there are fewer potential causes of degradation in forest lands than in agricultural or urban areas. Where there are fewer potential causes of degradation, historical reconstruction of habitat-forming processes may be the more direct approach because relatively few assessments can describe how processes and conditions have changed. Furthermore, the assessments can lead directly to spatially and temporally explicit conclusions about the causes of degradation, and can identify the appropriate actions for restoration. Thus, focusing on habitat-forming processes can be a more cost-effective approach to identifying restoration actions in forest lands or other areas with few potential causes of degradation.

Chapter 3. Disturbance and recovery definitions

Aquatic scientists generally accept that natural and anthropogenic disturbances are part of the aquatic ecosystem, and that recovery from individual disturbances can be described (e.g., over 150 case studies were reviewed by Niemi et al. 1990). However, the field of aquatic ecology has no theory that predicts the mosaic of aquatic conditions or ecological states caused by disturbances over extended time periods in the same way that theory predicts changes in terrestrial environments (Reeves et al. 1995). Thus, aquatic ecologists are unable to describe the changing patterns of habitat conditions or associated biota over the long term, or to predict recovery rates of aquatic ecosystems (Cairns 1990). Reasons for this lack of generally applicable theory include (i) the fact that many disturbance sequences are unique and may not be repeated (Cairns 1990), (ii) there is a wide array of space and time scales or physical and biological processes that may be included under "disturbance" (Gore et al.1990), and (iii) existing ecological theories may apply to only a subset of disturbance types when disturbance is defined too broadly (Gore and Milner 1990).

Research on ecosystem recovery after disturbance has touched on a wide variety of stressors and ecosystems in the past few decades, yet few of these studies can be considered similar to each other because they address almost as many different environments and indicators of recovery as there are studies (Niemi et al. 1990). This variation contributes to the difficulty ecologists have had in arriving at general theories of ecosystem recovery, but it does not entirely explain the lack of predictive capability in the field of aquatic ecology (Cairns 1990). Many disturbances that have been described in the fisheries and geomorphological literature are repeated over and over again (e.g., increased mass wasting due to logging or road building, decreased LWD recruitment due to riparian logging), providing ample opportunity to begin describing some common disturbance-recovery pathways that affect lotic habitats and biota.

Poorly defined space and time scales have also made it difficult to identify general ecological principles that shape recovery pathways (Allen and Hoekstra 1987). One of the most common problems encountered with respect to scale is that a disturbance at some smaller scale may not be viewed as a disturbance at a much larger scale. For example, at a landscape-level a single fire would not be considered a disturbance because it would not appreciably change the pattern of forest ages on the landscape. However, at the site-scale the same fire would be viewed as a disturbance to long-term stand development. Such differing views of the same event inhibit generalizations about disturbance and recovery of specific processes.

Finally, application of existing ecological theories to recovery pathways is made difficult by the fact that most ecological theories apply to a relatively narrow range of space and time scales, and to a relatively narrow range of disturbance types (Gore and Milner 1990). In some sense, this lack of generalized ecological recovery theory is a function of the tremendous variety of disturbance-recovery pathways and of poorly defined space and time scales. This suggests that perhaps an appropriate first step is to describe recovery pathways for a few specific habitat-forming processes at well defined space and time scales.

The terms *disturbance* and *recovery* have been defined in various ways by a number of authors in the past two decades, which has led to some confusion about their meaning as well as about the meaning of the related term *restoration* (Yount and Niemi 1990, Jordan 1995). Differences among many of the definitions result from differing spatial and temporal scales of interest (Fisher 1990, Allen and Hoekstra 1987, Poff and Ward 1990), and from differing types of influences on habitat or biota that researchers choose to include under disturbance (e.g., Reice et al. 1990). Such differences may legitimately preclude a general definition of these terms, which then forces those who use the terms to provide a specific definition. Here I do not provide an exhaustive review of the literature on disturbance and recovery, but instead summarize some examples of how the terms have been used and define how they are used in this dissertation.

Throughout the remainder of this dissertation I focus on describing recovery pathways for two specific processes: supply of sediment and LWD recruitment. These two processes have been chosen for study because changes in coarse sediment supply and woody debris abundance strongly affect salmonid summer and winter rearing habitats throughout the Pacific Northwest (e.g. Tripp and Poulin 1988a, Bilby and Ward 1991, Collins et al. 1994, Ralph et al. 1994, Beechie and Sibley 1997), as well as habitats for other life history stages such as spawning or incubation (e.g., Cederholm et al. 1981, Tagart 1984, Tripp and Poulin 1986b). Furthermore, degree of disturbance and recovery time have not been addressed for these processes as they have for other less complex problems of habitat loss such as isolation of existing habitats by culverts (Beechie et al. 1994, Beechie et al. 1996). Less complex disturbances have obvious causes, effects, and recovery time after restoration (e.g., replacement of a culvert that blocked upstream migration), and can be readily evaluated in the context of the restoration approach (Chapter 2).

By contrast. more complex processes such as sediment supply and LWD recruitment need more rigorous assessments of levels of disturbance and rates and pathways of recovery before they can be evaluated in the context of the restoration goal and priorities.

Disturbance

Disturbance has usually been defined as a deviation from some equilibrium condition or from a "normal" range of conditions (e.g., Pickett and White 1985, Resh et al. 1988), which is sometimes specific to biota (e.g., Poff and Ward 1990) and other times generally applied to biota and their environments (e.g., White and Pickett 1985). In cases where the disturbance is obviously outside the range of natural conditions (e.g., the introduction of a toxic substance with no natural analogue), relatively narrow definitions suffice because there is no need to account for natural variability or differences in spatial scale. However, when attempting to move toward a general *theory* of disturbance and its role in community organization, the definition must begin to account for natural processes that disrupt patterns in community structure (Reice et al. 1990). Selected definitions from recent literature illustrate how this problem has influenced changing definitions of disturbance in the past several years.

Milner (1994) chose to adopt the earlier definition of disturbance proposed by White and Pickett (1985), which is "any relatively discrete event in time that disrupts ecosystem, community or population structure and changes resources, substrate availability or the physical environment." This definition restricts a disturbance to short-term events, and considers virtually all changes in the environment or biota a disturbance. Other authors suggest that a dynamic environment is "normal", and that small events such as annual floods induce only small changes in the habitat or biota and should be considered normal (e.g., Resh et al. 1988, Reice et al. 1990). Reice et al. (1990) therefore chose to consider a disturbance regime which includes both predictable and unpredictable events. Predictable events are not considered disturbances because they are those events to which communities have adapted. By contrast, unpredictable events are considered disturbances because they are beyond the "normal" range to which communities have adapted and they cause significant changes in the biotic community. Finally, Sparks et al. (1990) accepted the preceding notion that disturbances must be outside some normal range of occurrences, but also considered that gradual changes to habitat or biota may be a disturbance and that disturbance should not be restricted to events that are discrete in time. They therefore chose to define disturbance as "an unpredictable, discrete *or gradual*, event (natural or manmade) that disrupts structure or function at the ecosystem, community, or population level."

Other authors have gone so far as to describe subsets of disturbance that are based primarily on the duration of the alteration to habitat or biota. Yount and Niemi (1990) adopted a distinction proposed by Bender et al. (1984) that disturbance could be subdivided into two temporally distinct types: *pulse* and *press* disturbances. Pulse disturbances are short-term and cause a relatively instantaneous change in populations, whereas press disturbances are long-term and produce a sustained alteration of certain species densities. Underwood (1994) further distinguishes *catastrophe* from press and pulse disturbances, suggesting that a catastrophic disturbance is a major alteration of habitat that is permanent and that causes a permanent alteration of species densities.

For most of these definitions there is an assumption that a "normal" condition exists, which can also be described as the "nominal behavior" of a system (Yount and Niemi 1990). Furthermore, the use of these definitions in a management setting (e.g., comparing disturbed to undisturbed conditions) implies that the "normal" condition can be measured or estimated, and that it can be used as a reference condition with which to compare disturbed conditions. However, it is worth noting that other authors argue that natural variability may preclude the definition of a "normal" condition (e.g., Rice 1982, Kelsey 1982), and that there is no long-term equilibrium state against which to measure change. This is based primarily on the contention that the interval between disruptive events is shorter than the recovery time, which suggests that conditions are constantly in flux and that there can be no measurement which represents "normal". For processes where this is the case, definitions of disturbance that are based on a "normal" or reference condition become invalid. However, definitions that include only events that are relatively discrete in time and that disrupt the physical environment or ecosystem structure (e.g., White and Picket 1985) may remain useful.

Given the preceding differences among definitions of disturbance, it can be argued that any definition of disturbance is both spatially and temporally scale dependent (Allen and Hoekstra 1987, Poff and Ward 1990). It follows then that some events might be considered a disturbance when viewed at a site, but as normal behavior of the system when viewed across a landscape. For example, at the forest plot scale (10^1-10^2 m^2) a stand-replacing wildfire may occur only once every few hundred years (Heyerdahl et al. 1995) and must be considered a disturbance because it completely disrupts what is otherwise a slowly and continually developing forest. However, at the landscape level ($\geq 10^4$ km^2), one will observe at least one fire every few years and may observe several fires in some years (Benda 1994). From this point of view the observer would be inclined to consider fire a normal process that creates a patchy forest landscape, and a single fire would not be considered a disturbance because it would not appreciably change the pattern of forest ages on the landscape.

For the purposes of this dissertation, I define disturbance as a significant change in the rate or magnitude of a salmonid habitat-forming process. The two examples considered here are supply of sediment and supply of large woody debris (LWD) to stream channels. A variety of measures of disturbance may be used depending on spatial scale of interest and on the environmental conditions of interest (Figure 3-1). Indicators of disturbance may describe (i) factors that cause changes in rates or magnitudes of processes, (ii) the rates or magnitudes themselves, or (iii) the effects of changes in rates or magnitudes of processes. For this dissertation, indicators of disturbance may describe factors that cause changes in the rates of sediment supply and LWD recruitment, changes in the rates of sediment supply and LWD recruitment themselves, or the channel or habitat features affected by changes in those rates. Disturbances to these two processes are commonly caused by fire in the natural forested environment and by logging or road building in the managed forest environment. At the site level, indicators of disturbance will typically be a quantitative measure of some environmental feature such as stand age, number and size of trees in a stand, LWD abundance in a reach, or pool spacing. Landscape-level indicators of disturbance will most often be an aggregate of site level measures, which can be considered a description of the "disturbance regime". A disturbance regime can be expressed as a frequency distribution of the types of site level indicators described above (i.e., causal factors, process rates, or conditions). Examples of these might be the frequency distribution of time since the last fire at many points across a landscape (Benda 1994), the frequency distribution of number of landslides per unit time (Benda and Dunne 1997a), or the frequency of occurrence of bed material thickness in a channel network (Benda and Dunne 1997b).

The meaning of the term significant can be left vague in the general definition of disturbance because its meaning is dependent first on the indicator of disturbance or recovery that is chosen (Kelly and Harwell 1990), and second on the spatial and temporal scale at which a process is

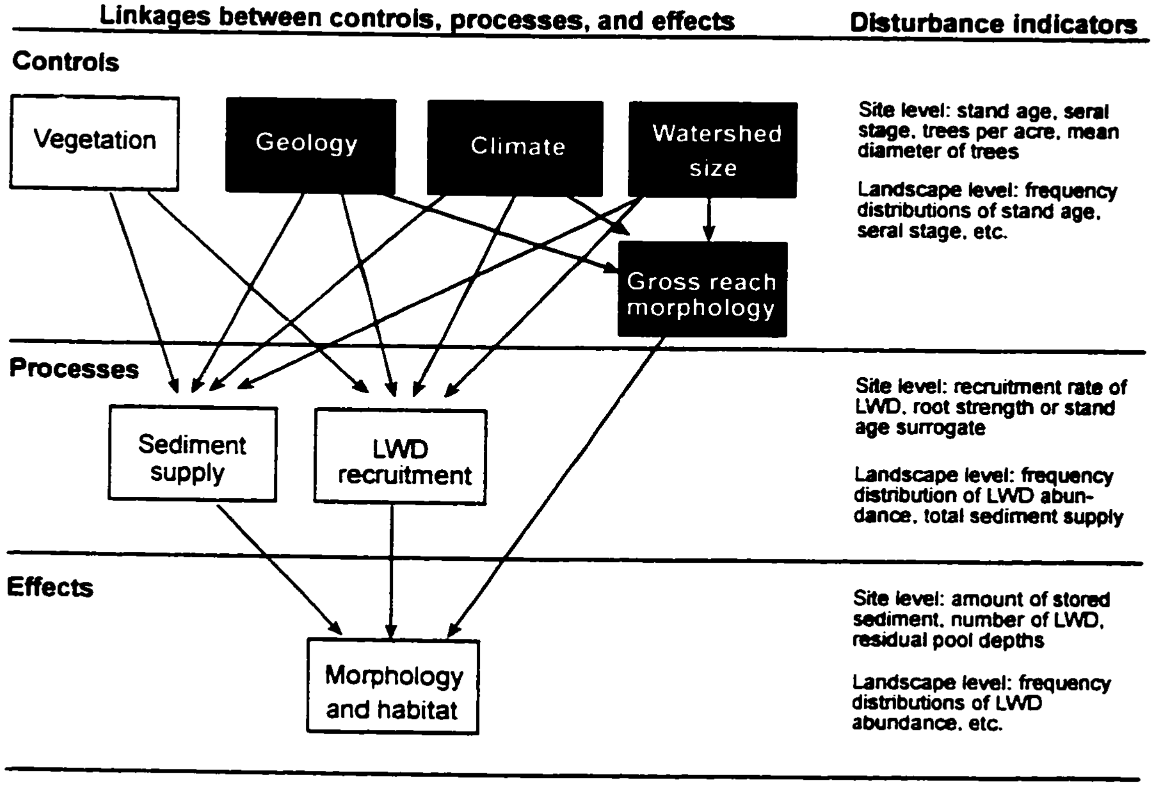


Figure 3-1. Site-level indicators of disturbance may describe (i) factors that control rates or magnitudes of processes, (ii) the rates or magnitudes themselves, or (iii) the effects of changes in rates or magnitudes of processes. Measures of disturbance at the landscape level are frequency distributions of site level disturbance indicators. Black boxes indicate those controls that are not affected by land use over the time frame considered in this study (10^1 years). Gross reach morphology refers to average channel slope, approximate size of the channel, and flood plain width.

viewed (Allen and Hoekstra 1987, Sedell et al. 1990, King 1993). In general, however, I will interpret significance based primarily on the effect of changes in process on the characteristics or function of the physical stream system. In the context of the restoration approach presented in Chapter 2, the most important aspect of disturbance is the degree to which it affects habitat conditions. For different locations and processes, comparisons of degrees of disturbance help determine which disturbances are the most important causes of habitat degradation, and which restoration actions will likely produce the greatest improvement in a locally selected biological indicator.

Recovery

Not surprisingly, defining the term recovery suffers from many of the same problems as defining the term disturbance. Some aspects of this confusion are described by Fisher (1990), whose discussion of succession encapsulates many of the problems associated with the term recovery. Fisher implies that the search for a generalizable theory of recovery is futile because it has few generalizable features applicable to a range of ecosystem types. Instead, he argues that recovery is but one aspect of "ecosystem stability", and that research into ecosystem recovery must be conducted in this broader context. However, other authors feel it is not unreasonable to expect the definition of recovery to be flexible if the term is to be applied across a wide variety of space and time scales (Gore et al. 1990).

Reasons for confusion about the term recovery are many, including (i) insufficient understanding of how organisms interact with each other and their environment both before and after disturbance (Fisher 1990), (ii) varying definitions of recovery as the space and time scale of interest changes (Gore et al. 1990), and (iii) the unique and unrepeated nature of many specific disturbances (Cairns 1990). In large part, these obstacles are created by our lack of understanding of how a complex, hierarchically-organized lotic ecosystem works (Allen and Hoekstra 1987), and by the lack of a clearly defined "normal" condition from which to measure change (Reice et al. 1990, Benda 1994, Milner 1994). On the basis that "normal" is often impossible to identify, Milner (1994) suggests that recovery is best described as "a return to an ecosystem which closely resembles unstressed surrounding areas" (Gore 1985) or to "an unstressed condition" (Kelly and Harwell 1990). Others suggest that the term recovery must be defined for a specific space and time scale, which should be relatively straight forward when the system is viewed as a hierarchy of space and time scales (Allen and Hoekstra 1987, Poff and Ward 1990).

The term recovery as applied to lotic systems more often refers to biota than to the environment that supports them (e.g., Gore et al. 1990, Niemi et al. 1990, Yount and Niemi 1990). Where recovery has specifically referred to fishes (i.e., to first reappearance of a species of fish, to a pre-disturbance fish density, or to a pre- disturbance individual size), the time span of recovery once a disturbance is removed is generally less than three years (Niemi et al. 1990). However, when the habitat was modified or isolated from a source of colonists to repopulate the area, recovery time for fishes generally followed that of the environment. Therefore, where there is a widespread and long-term modification of the environment, one might presume that biota will not recover until the environment recovers, and managers might logically focus restoration efforts on the environment rather than biota.

The assumption that habitat must recover before biota can recover presently drives much of the habitat restoration in the Pacific Northwest. This assumption is supported by research showing increases in local fish abundance as a result of habitat modifications (e.g., House and Boehne 1986, Cederholm et al. 1988, Nickelson et al. 1991). Summer use of constructed pools by juvenile coho salmon increased significantly when wood or rock structures were added Oregon coastal streams (Nickelson et al. 1992), and populations of juvenile salmon and trout increased significantly when woody debris structures were added to a stream reach in Tobe Creek, Oregon (House and Boehne 1986). Juvenile coho salmon were also more numerous and grew faster during winter in constructed off-channel habitats (Nickelson et al. 1988, Cederholm et al. 1991). Others have built upon this and other research to develop predictive tools that estimate fish abundance based on physical characteristics of habitat (e.g., Marshall et al. 1980, Reeves et al. 1989, Lestelle et al. 1993) and analytical tools that describe changes in fish production caused by changes in habitat conditions (Beechie et al. 1994, Collins et al. 1994).

In keeping with the goal of restoring habitat-forming processes (Chapter 2), I define recovery at the site or reach level (10^{-2} to 10^{0} km^2) as a return to the natural rate or magnitude of habitat-forming processes (Figure 3-2a). This definition of recovery is adequate for threshold processes that can be considered either disturbed or recovered at any point in time. However, in some cases the post-disturbance "recovery" is gradual and long-term, and a process may not stabilize between disturbances (Figure 3-2b). In these cases, a gradual recovery pathway can be

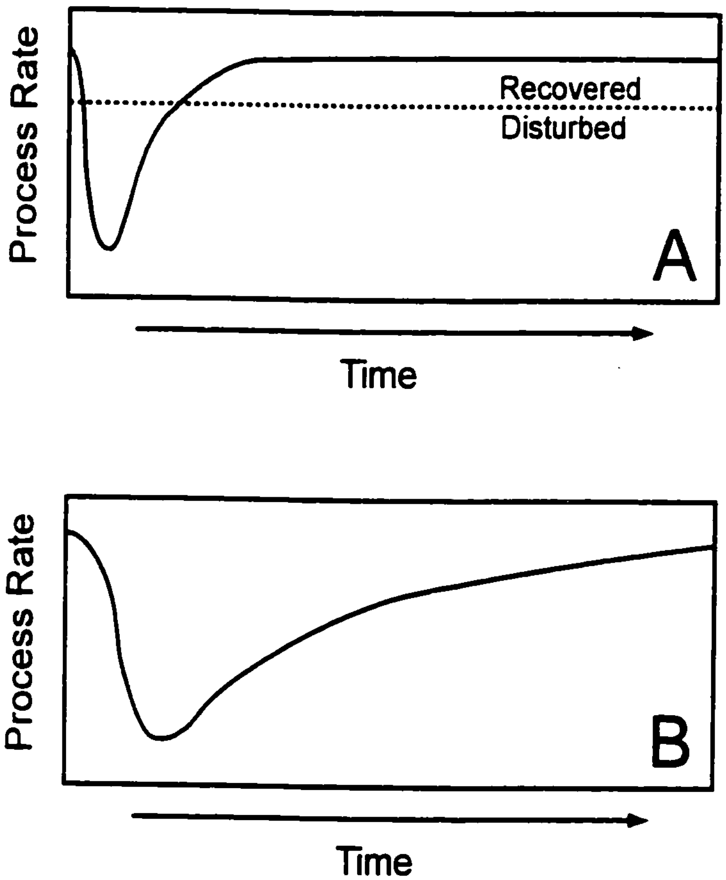


Figure 3-2. Plot of the disturbance-recovery cycle for a habitat-forming process at a single site that (A) has a recovery threshold, or (B) varies continuously with time since disturbance and may not stabilize between disturbances.

described and the definition of recovery may be modified to include "changes that occur on a site after disturbance" (Fisher 1990). (One example of such a pathway is the development of riparian forest and associated changes in amount and size of LWD recruited to a stream.) In general, disturbance at a site will refer to (i) a change in a factor that influences the rate or magnitude of a process, (ii) a change in the estimated rate of a process, or (iii) a change in a habitat condition that is affected by an altered process (see Figure 3-1). Where there is a threshold of recovery, a process (or indicator of its cause or effect) will be considered "recovered" when it returns to the "natural" rate. When there is no threshold of recovery, recovery simply refers to the pathway that a selected indicator follows after disturbance.

At a landscape-level (spatial scale $\geq 10^3$ km^2), a "disturbance regime" can be described as change in the type or frequency of site-level disturbances. Therefore, recovery can be defined as a return to the natural frequency and magnitude of those disturbances. However it is not possible to describe recovery pathways at the landscape-level in the same way that recovery pathways can be described for site-level disturbances. Rather, a landscape-level recovery pathway for sediment supply or LWD recruitment is the aggregate of site-level recovery pathways.

Chapter 4. Disturbance and recovery of sediment supply processes

Changes in forest cover produced by fire or forest management change the rates of sediment supply to streams (e.g., Sidle et al. 1985, Cederholm et al. 1982), which alter the characteristics of salmonid habitats (e.g., Tripp and Poulin 1986a, Tripp and Poulin 1986b), which then change the salmonid carrying capacity or productivity of a stream (e.g., Hicks et al. 1991, Collins et al. 1994). This chain of cause-effect linkages ties the influences of fire and land uses on sediment supply to fish habitats and fish production in a channel network. By understanding these linkages, we can begin to identify *causes* of habitat degradation, which is a step toward the restoration of processes that form habitats to which salmonids are adapted (Frissell and Nawa 1992, Roper et al. 1997, Beechie et al. 1996).

Clearcut logging and stand-replacing fires in mountainous areas can lead to large increases in sediment supply to streams, primarily as a function of root strength decay and the resultant increased potential for landsliding (O'Loughlin 1974, Burroughs and Thomas 1977, Sidle et al. 1985). Mass wasting rates in clearcuts typically increase by a factor of two to three west of Cascade Mountain crest in Oregon, Washington, and British Columbia (Sidle et al. 1985). The rate of increase after fire in the same area is not well known, but anecdotal evidence of increased mass wasting after fire in the North Cascades supports the contention that fire increased rates of mass wasting in the study area (Ayres 1899). Forest roads can indirectly cause landslides downslope of the road location by concentrating water into mass wasting-prone areas (Sidle et al. 1985, Montgomery 1994). Failures also occur relatively frequently on oversteepened cut and fill slopes (Sidle et al. 1985), and where woody debris buried in the prism of older roads decays and causes failure of the road prism. Mean increases in mass wasting rate due to road construction ranged from 30 to 346 times mass wasting rates in mature forests in nine different studies in the Pacific Northwest (Sidle et al. 1985).

An increase in supply of sediment to a reach has two main effects on channel morphology that are linked to salmon productivity in the Pacific Northwest. The first is the influence on pool abundance and depth, and the second is the influence on channel geometry. The ratio of sediment supply to sediment transport capacity strongly influences pool abundance and depth (Lisle 1982, Collins et al. 1994), in turn affecting the quantity and quality of rearing habitat for juvenile salmonids or holding habitat for adult salmonids (Tripp and Poulin 1986a, Reeves et al. 1989, Collins et al. 1994). Changes in channel geometry caused by increased sediment supply are also important in that they indirectly affect other aspects of habitat conditions such as stream temperature (e.g., Collins et al. 1994) or survival of salmonid eggs in spawning gravel (e.g., Tripp and Poulin 1986b).

There is often a time lag of years to decades between a change in sediment supply and a change in channel morphology of a reach downstream in the channel network (e.g., Kelsey 1982b, Madej and Ozaki 1996). This lag is due to the time required for sediment to travel from its source to the reach of interest (Kelsey 1982a). Once sediment is deposited in a reach, its persistence is a function of the sediment transport capacity of the reach. Therefore, both the timing and persistence of changes in the morphology of downstream reaches are functions of the rate at which sediment moves through a channel network.

This chapter separates the effects of changing sediment supply on salmonid habitat into two parts. The first part is the effect of land use on sediment supply, and the second part is the effect of altered sediment supply on channel morphology. The first part of the problem is largely described in another component of this project: *Estimating changes in sediment supply due to forest practices* (Paulson 1997). The first section of this chapter describes land use effects on sediment supply at both site and landscape scales, and describes the magnitude and duration of changes in sediment supply due to land use at both scales based on Paulson (1997). The second section discusses effects of changing sediment supply on channel morphology, which is based partly on a second component of this project: *Effects of sediment supply and LWD abundance on fish habitat characteristics* (Nelson 1998). It reviews the types of effects that sediment supply has on channel morphology and summarizes Nelson's (1998) estimates of the degree to which varying levels of sediment supply affect habitat characteristics. The remainder of the second section focuses on estimating the duration of effects of altered sediment supply, which is essentially a problem of estimating rates of sediment movement through stream channels. It describes a simple method of estimating the rate of sediment movement through a reach and gives examples of the application of this methodology.

Magnitude and duration of land-use effects on sediment supply

This section describes the magnitude and duration of land-use effects on sediment supply at site and landscape scales. The section begins with a discussion of site-level effects of logging and road construction on mass wasting. Its purposes are to describe (1) the physical mechanisms by which land uses alter sediment supply processes, and (2) the magnitude and duration of these disturbances. The section then describes how site-level effects are expressed at the landscape level under different disturbance regimes. Two wildfire scenarios represent the "natural" disturbance regime, and three forest management scenarios represent present conditions. Comparisons among disturbance regimes are expressed in terms of average annual sediment supply per unit area of watershed.

Fine sediment (≤ 2 mm intermediate diameter) is a product of both surface erosion and mass wasting processes, and it has effects on both spawning and rearing habitats. However, increased supply of fine sediment to spawning gravels in the Skagit River study area does not appear to be a factor that limits the production of coho salmon (Beechie et al. 1994), nor does it appear to be a principle factor in the decline of other nearby salmonid stocks such as the Deer Creek summer-run steelhead trout (Collins et al. 1994). Because increased fine sediment supply does not appear to be a factor that limits salmonid production in the study area and surface erosion produces almost exclusively fine sediment (Reid et al. 1981), this chapter does not discuss the effects of surface erosion on sediment supply. Instead, this chapter focuses on supply of coarse sediment (> 2 mm intermediate diameter), which is produced primarily by mass wasting processes and has significant effects on salmonid rearing habitats. Such changes to rearing habitats appear to be one factor affecting abundance of coho salmon and steelhead trout in the North Cascades (Beechie et al. 1994, Collins et al. 1994). Surface erosion processes are discussed in Paulson (1997).

Site-level disturbance and recovery of sediment supply processes

Supply of coarse sediment (> 2 mm) to mountain stream channels is moderated by the presence of dense forests, primarily because roots reinforce soils in mass wasting-prone areas (Sidle et al. 1985, Benda 1994). Mass wasting rates tend to increase after fire or harvest due to decaying roots of dead trees and then return to pre-disturbance levels as new trees are established (Sidle et al. 1985). At the site level, dead roots decay rapidly at first, losing approximately 60% of their strength in the first 5 years (Figure 4-1a). Reinforcement of soils by roots of dead trees is about 10% of that of the pre-clearcut stand after 10 to 15 years. However, reinforcement by roots of new trees compensates for a portion of the decay of dead roots, recovering as much as 70% of the original rooting strength within as little as 12 years. The lowest total rooting strength occurs approximately 7 years after clearcutting, suggesting that landslide rates should be highest around 7 years after clearcutting or fire. This is supported by studies in the Pacific Northwest, Japan, and New Zealand that have shown increased landslide rates between 3 and 10 years after clearcutting (Sidle et al. 1985). Root strength can be assumed to be substantially returned to pre-clearcut levels after approximately 20 years because empirical evidence does not suggest increased mass wasting rates after that time (Sidle 1985).

Recovery of root strength after wildfire or clearcutting may take longer than 20 years if forest regeneration is poor. Anecdotal evidence in the North Cascades indicates that some areas had poor regeneration for many years after a fire, whereas others regenerated rapidly (Ayers 1899). It was also noted, however, that there had been "no fires of magnitude" in the study area at the turn of the century (Gannett 1899), and that less than 2% of forests were considered unmerchantable due to fire at that time. Because there appeared to be relatively few fire-disturbed areas and few areas with poor regeneration, I assumed that post-fire recovery of root strength is similar to that after logging (i.e., 20 years).

The change in relative mass wasting rate after a disturbance is presumed to correlate with a change in root strength, increasing to a maximum rate about 7 years after disturbance (dotted curve in Figure 4-1b). (Relative mass wasting rate is the mass wasting rate for stands ≤ 20 years old divided by the mass wasting rate for stands > 20 years old.) However, rates of mass wasting are typically assessed using aerial photographs taken at intervals of several years, making estimation of changing mass wasting rates on a yearly basis impossible. For this study, changes in mass wasting rate are therefore estimated as the average rate for either "disturbed" or "recovered" areas. Where stand age is ≤ 20 years, the site is considered to be disturbed and to have an increased likelihood of mass wasting (Figure 4-1b). Where stand age is > 20 years, the site is considered to be recovered and to have a "normal" likelihood of mass wasting. Relative mass wasting rates in nine sub-basins of the Skagit River range from 1 to 9, with a median of 4 (Paulson 1997). There was no relationship between the relative mass wasting rate and the mature forest rate (stand age > 20 years), indicating that the effects of land use were not a function of rates of mass wasting in mature forest. However, the average mature forest rate for individual sub-basins was strongly influenced by local geology and landforms (Paulson 1997).

Forest roads affect mass wasting rates through a variety of mechanisms that have less predictable recovery times than does root strength. Road construction can indirectly cause

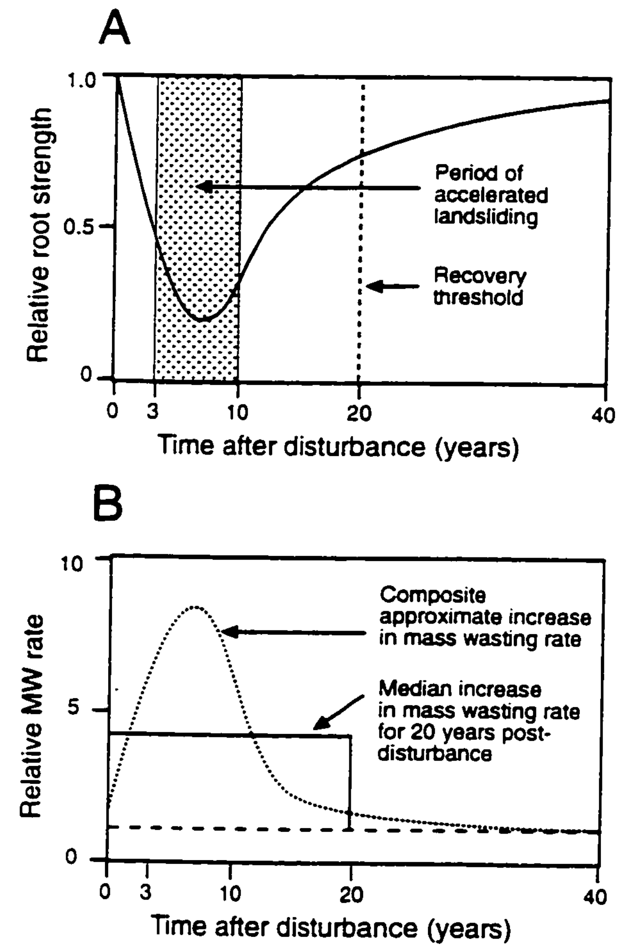


Figure 4-1. (A) Generalized decline and recovery of total relative rooting strength in a soil after clearcutting (adapted from Sidle et al. 1985). Empirically, increased incidence of landsliding tends to occur between 3 and 10 years after clearcutting (gray area). Root strength is assumed to be substantially returned to pre-harvest or pre-fire levels after approximately 20 years (Sidle et al. 1985). (B) Generalized (composite) increase in mass wasting rates after a disturbance, peaking approximately 7 years after disturbance (dotted line) (based on Sidle et al. 1985 and Paulson 1997). The "normal" or average relative rate of mass wasting for an area with vegetation >20 yrs old is 1 (dashed line), and "disturbed" rates are expressed in multiples of the average rate. The average rate of mass wasting during the first 20 years after clearcutting in the North Cascades of Washington was 4 times the rate for forests > 20 years old (solid line) (Paulson 1997).

landslides downslope of the road location by concentrating water into mass wasting-prone areas (Sidle et al. 1985, Montgomery 1994). Although most current road designs attempt to avoid such failures, they remain relatively common due to planning errors and due to failure of drainage structures. Failures also occur relatively frequently on oversteepened cut and fill slopes (Sidle et al. 1985), and where woody debris buried in the road prism of older roads decays and causes failure of the road prism. Both new and old roads experience landslides (Toth 1991, Harr and Nichols 1993), and road-related landslide rates in 9 sub-basins of the Skagit River remained constant over time (based on data in Paulson 1997). Thus, there appears to be no true recovery for the influence of roads on the mass wasting process, and road areas are considered to be constantly in the disturbed state regardless of road age. Paulson (1997) showed that road-related relative mass-wasting rates (mass wasting rate for roads divided by the mass wasting rate for mature forest) in nine sub-basins of the Skagit River ranged from 6 to 1767, with a median of 45. This value is consistent with that of other studies, where relative mass wasting rates due to road construction ranged from 30 to 346 (Sidle et al. 1985).

Landscape-level disturbance and recovery of sediment supply processes

This section first describes how site-level disturbances to mass wasting processes can be aggregated to describe a landscape-level disturbance regime. It then describes the estimation of stand age distributions across the landscape under two disturbance regimes: a natural disturbance regime with fire as the dominant disturbance mechanism, and a forest management disturbance regime with logging and road building as the dominant disturbance mechanisms. The fire descriptions draw on various sources of fire frequency information and descriptions of forest conditions prior to widespread logging and other land uses in northwest Washington (Appendix A). The stand age distribution for managed forests is based primarily on local timber harvest practices such as rotation length and proportions of the landscape in roads. Finally, these distributions are applied to sediment supply processes in order to describe a generalized disturbance and recovery pattern for landscape-level sediment supply.

A landscape-level change in "disturbance regime" is simply a change in the frequency or magnitude of site-level disturbances. For mass wasting, land-use disturbances at the site-level are caused by logging or road construction, which increase average mass wasting rates by factors of 4 and 45, respectively (Paulson 1997). Mass wasting rates after logging recover to approximately pre-disturbance levels when stand age reaches about 20 years, whereas as effects of road construction have no apparent recovery. For the purposes of this study, mass wasting rates after fire are assumed to have a magnitude and duration that is similar to that of clearcut areas. Knowing the magnitudes and durations of these disturbances, it is then possible to describe a disturbance regime for mass wasting by estimating the proportions of the landscape that have increased average rates of sediment production due to fire, logging, or road construction. Because mass wasting rates for burned and clearcut areas recover after about 20 years, the important data are the proportions of the landscape that remain below the 20-year recovery threshold. For roads, however, age is unimportant and only the total area of roads is required. With these data, it is possible to identify a change in disturbance regime at the landscape-level, and to estimate the relative effect such a change has on average annual sediment supply.

The foremost assumption in describing the natural disturbance regime for sediment supply is that fire was the primary vegetation disturbance mechanism prior to Euroamerican settlement. The first reason for this assumption is that fire is a dominant factor affecting sediment supply in the natural environment of the Pacific Northwest (Benda 1994), and sediment supply has significant effects on salmonid habitat (Hicks et al. 1991). The second reason is that the appropriate comparisons between present conditions and historical conditions must include those factors that affect similar processes. Because logging practices are presently the dominant cause of increased mass wasting rates in the study area (see Paulson 1997), comparisons of the natural sediment supply regime to the present sediment supply regime should consider fire the disturbance mechanism in the natural regime and clearcut logging the disturbance mechanisms in the forest management regime. Although variations in storm size and intensity may also cause large fluctuations in sediment supply, storms are not included as a disturbance mechanism because they are unaffected by changes in the vegetation disturbance regime (see Figure 3-1). Variation in sediment supply due to storms under the forest management disturbance regime is assumed comparable to that under the natural disturbance regime because changes in mass wasting rates due to variation in storm magnitude should be random (Benda 1994).

At the landscape scale, a distribution of stand ages can be used to estimate the proportion of a landscape that is below the recovery threshold for root strength (stand age ≤ 20 years) for either disturbance regime. Areas of the landscape below the recovery threshold have higher mass wasting rates than areas above the recovery threshold (Paulson 1997), indicating that a change in the proportion of the sub-basin or landscape below the threshold results in a change in the total sediment supply. However, variations in storm size and intensity may cause fluctuations in sediment supply that are greater than those caused by fire or land use (Benda 1994). Nevertheless, effects of land use on sediment supply should be expressed during both small and large storms, meaning that the entire distribution of sediment supply values should be shifted toward larger supply as the proportion of the landscape with stand age less than 20 years increases. Thus, over a long enough time period, a shift in the distribution of site-level rates of sediment production can be described by a change in the long-term average rate of sediment production (e.g., Paulson 1997).

Under the natural fire regime, the mean proportion of stand ages less than 20 years is 5% for the silver fir zone ($1-e^{-(.0028)(20)}$), and 10% for the western hemlock zone ($1-e^{-(.005)(20)}$) (Figure 4-2, Appendix A). However, natural variability in fire inter-arrival times can lead to significant variation in the percentage of stands less that 20 years old at any one point in time. I estimated this variability by a Monte Carlo simulation of the inter-arrival times of fires in the silver fir zone for 100 sites and over 5000 years using a log-normal distribution of inter-arrival times (Appendix A). Over a 2000-year segment of that 5000-year modeling period, sampling the percentage of silver fir stands younger than 20 years every 100 years showed that the percentage varied from 1% to 11% due to random variation in times between fires (Figure 4-2). For the western hemlock zone, I assumed that variation in the percentage of stands less than 20 years old is proportional to that in the silver fir zone because the form of the driving equation in a Monte Carlo simulation is identical for both stand types. Thus, I estimated that the minimum and maximum percentages for the western hemlock zone were two times higher than those of silver fir (based on the fact that the western hemlock average percentage was twice that of silver fir), yielding estimated minimum and maximum percentages of stands <20 years old at 2% and 22%, respectively.

For the forest management disturbance regime, the mean percentage of stands younger than 20 years under a 60-year harvest rotation is 33%, and forest roads typically occupy about 2% of the watershed area (Figure 4-2). Varying the mean harvest rotation from 40 to 80 years results in the percentage of stands less than 20 years ranging from 25% to 50%. Even at the longer rotation age of 80 years, the proportion of stands younger than 20 years (25%) remains higher than the estimated maximum percentage for the western hemlock fire regime (22%). The proportion of

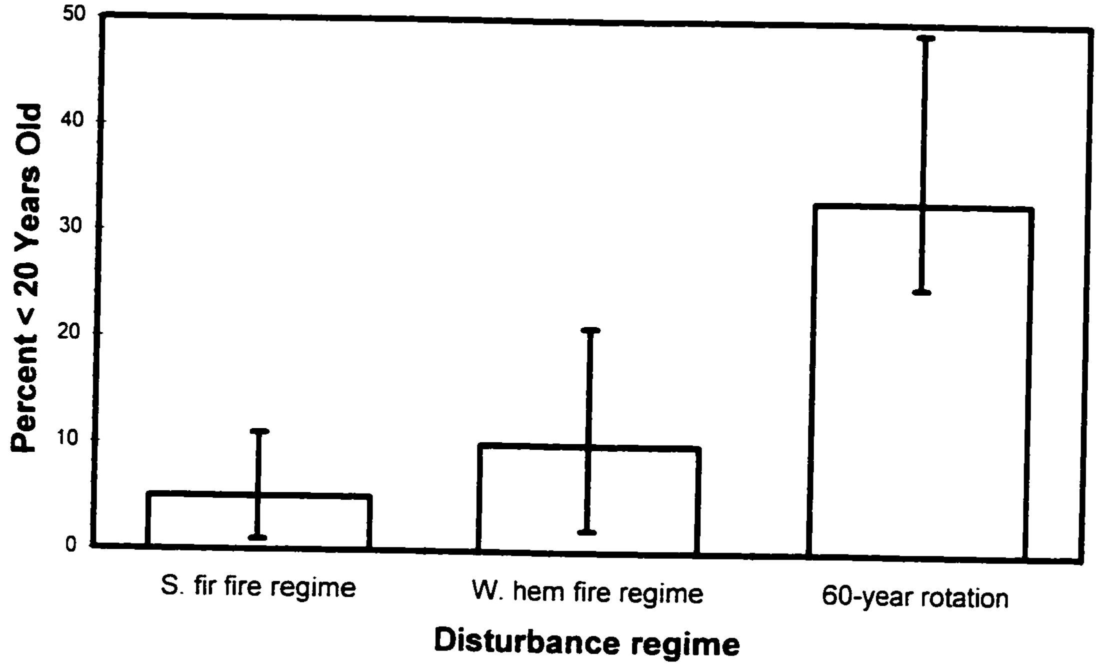


Figure 4-2. Percent of stands less than 20 years old under the silver fir fire regime, the western hemlock fire regime, and a 60 year harvest rotation. Error bars for silver fir fire regime estimated minimum and maximum percentage from modeling of 100 stands over a 2000-year period. Error bars for western hemlock fire regime are proportional to those for silver fir. Error bars for the 60 year timber harvest rotation represent alternate rotations of 40 years (upper) and 80 years (lower).

stands younger than 20 years can thus be more than three times higher than under the natural fire regime when mass wasting-prone areas are not protected from timber harvest, as has been the case until the last decade.

With young stands of the study area producing sediment at an average rate 4 times higher than that of mature forests (see preceding section and Paulson 1997), changing the percentage of stands below threshold from 10% to 33% would result in an estimated 53% increase in sediment supply across the landscape. In addition to timber harvest effects on sediment supply, the average rate of sediment production from road-related mass wasting is 45 times that of mature forest areas (see preceding section and Paulson 1997). Road construction at a level of 2% of the watershed area combined with the 60 year harvest rotation would then result in an estimated 121% increase in sediment supply from mass wasting across the landscape (Table 4-1). Variation in percent of stands below threshold for the forest management regime based on the 40-year and 80-year harvest rotations (from 25% to 50% of the area less 20 years old) and in percent road area (from 1% to 3%) could produce sediment at average annual rates ranging from 70% to 300% higher than under the natural fire regime (Table 4-1).

Recovery to the natural distribution of stand ages in commercial forests would, of course, take centuries. With very few stands greater than 100 years old, recovery of a significant portion of the landscape to greater than 300 years of age is more than 200 years away. However, from a mass wasting perspective, recovery of logged areas is little more than 20 years away with severe harvest reductions (as has happened on federal forests in the study area), and perhaps not more than a few decades with less severe measures. At present, mass-wasting-prone areas on state and private lands can be protected from harvest to reduce the effect of forest management on mass wasting (e.g., Collins et al. 1994, Washington Department of Natural Resources 1994). In this way, longer harvest rotations are targeted to those areas that have a significant impact on salmonid habitats. This "passive restoration" technique may significantly reduce the proportion of mass-wasting-prone areas with stands younger than 20 years, and may reduce sediment supply in the study area by an average of 47% (Paulson 1997). Furthermore, such efforts would typically be expected to occupy <20% of the watershed area (Paulson 1997).

No recovery of mass wasting rates from roads is expected without management intervention. (Recall that there were no discernible trends in mass wasting rates from roads as a function of time or as a function of new road construction [Paulson 1997].) Two types of active intervention

Table 4-1. Relative rates of sediment supply under typical natural and management disturbance regimes. The base rate is the average sediment production rate for the western hemlock zone with the long-term average percentage of stands <20 years old. Stands less than 20 years old have a relative sediment production 4 times that of stands >20 years old, and roads have a relative sediment production rate 45 times that of stands >20 years old (Paulson 1997).

Disturbance regime	Relative sediment production rate
Silver fir fire regime	
Minimum percent <20 years old (1%)	0.79
Long-term average percent <20 years old (5%)	0.88
Maximum percent <20 years old (11%)	1.02
Western hemlock fire regime	
Minimum percent <20 years old (2%)	0.82
Long-term average percent <20 years old (10%)	1.00
Maximum percent <20 years old (22%)	1.28
80-yr harvest rotation (25% <20 years old)	
Minimum road density (1%)	1.68
Average road density(2%)	2.02
Maximum road density (3%)	2.36
60-yr harvest rotation (33% <20 years old)	
Minimum road density (1%)	1.87
Average road density(2%)	2.21
Maximum road density (3%)	2.55
40-yr harvest rotation (50% <20 years old)	
Minimum road density (1%)	2.26
Average road density(2%)	2.60
Maximum road density (3%)	2.94

are possible: reconstruction to higher standards or removal of hazardous roads. Reconstruction to higher standards would typically include design of stream crossings to accommodate larger floods, design of crossings so that culvert blockages do not initiate landslides, and full-bench construction where side-cast failure can deliver to streams. Road removal would typically include sidecast pull-back, culvert and bridge removal, and re-establishing surface drainage pathways. Both methods are being actively employed throughout the Pacific Northwest, although there appears to be no documentation of their effects on sediment supply due to relatively recent implementation. Monitoring of both practices is underway in the study area, although the effectiveness of these techniques will not be known until after at least one decade of monitoring (Beamer et al. 1997).

Magnitude and duration of effects of altered sediment supply on stream channels

This section is separated into three parts: (1) the generalized disturbance-recovery cycle for changing sediment supply in a stream reach, (2) the processes by which changing sediment supply alters channel morphology, and (3) the processes by which a channel recovers from increased sediment supply. The first part explains in general terms how changes in sediment supply affect channel geometry and morphology, as well as the processes by which recovery occurs. The second part provides greater background on the mechanisms responsible for changes in channel morphology, focusing on those mechanisms that are related to the influence of sediment supply on pool depth. The third part describes sediment transport through stream channels. It first reviews some of the literature on sediment transport, focusing on sediment transport mechanisms that influence the rate of sediment movement through a stream reach. It then develops an index of sediment transport rate for use in estimating recovery time.

Disturbance-recovery cycle for channel responses to increased sediment supply

The sediment disturbance-recovery cycle for a stream channel in western Washington is typically characterized by a rapid increase in sediment supply relative to transport capacity, followed by a gradual export of sediment from the reach. An increase in sediment supply can, of course, range from moderate to severe, and the effects on a stream channel may range from minor aggradation to valley floor burial (Harvey 1987). These levels of effect may be described simply by a classification scheme (Harvey 1987), or by more specific measurements such as changes in quantities of channel stored sediment (Madej 1992) or changes in pool depths (Collins et al. 1994). Here I describe levels of sediment supply in terms of quantities of stored sediment in the channel bed, bars, and flood plain (Figure 4-3). Each of the levels of increased sediment reflect oversupply of sediment relative to the transport capacity of a reach. *Moderate* increases in sediment supply are characterized by increased storage of sediment in the bed and bars, with relatively small changes in channel width. Localized fining of the bed surface and pool filling are also likely with moderate increases in sediment supply. *High* sediment supply is characterized by a dramatic increase in the amount of sediment stored in bars, a portion of which is usually recruited from the flood plain storage reservoir. That is, increased bar sizes are partly due to increased sediment supply from upstream and partly due to channel widening and recruitment of flood plain sediments. Fining of the bed surface is more widespread and pools are substantially filled when sediment supply is high. *Extreme* sediment supply is characterized by lack of a defined channel, aggradation to at least flood plain level, and severe channel widening. The lack of defined channels characterizing extreme sediment supplies may be a short-lived feature of aggraded channels.

Definition of storage reservoir boundaries is somewhat arbitrary. In this study, flood plains are defined as channel-adjacent flat surfaces with vegetation ages exceeding five years, and which appear to be flooded relatively frequently. and poorly defined overflow channels. In most reaches of the study area flood plains are between 0.7 m and 1.5 m above the channel bed at riffles. Channel adjacent terraces are higher than flood plain elevation, and do not have indications of frequent flooding.

The channel bed and bars have vegetation ages less than 5 years, based on Madej (1992). Bars are topographic highs within the channel, usually with surface grain sizes much smaller than those of the adjacent channel bed. Grasses, shrubs, and small trees may grow on the surfaces of higher bars. The boundary between bars and the channel bed is indistinct, although there is often an identifiable change in grain size and cross-channel slope. The location of the bar-bed boundary is often found near the summer low-flow water surface, and in the field use of the water surface as the boundary appears to be adequate. Bar heights (measured from riffle beds) range from a few tenths of a meter to the height of the flood plain.

A hypothetical cycle of disturbance and recovery describes characteristics of a channel at a cross section as a sediment wave passes through (Figure 4-4), based primarily on Madej and

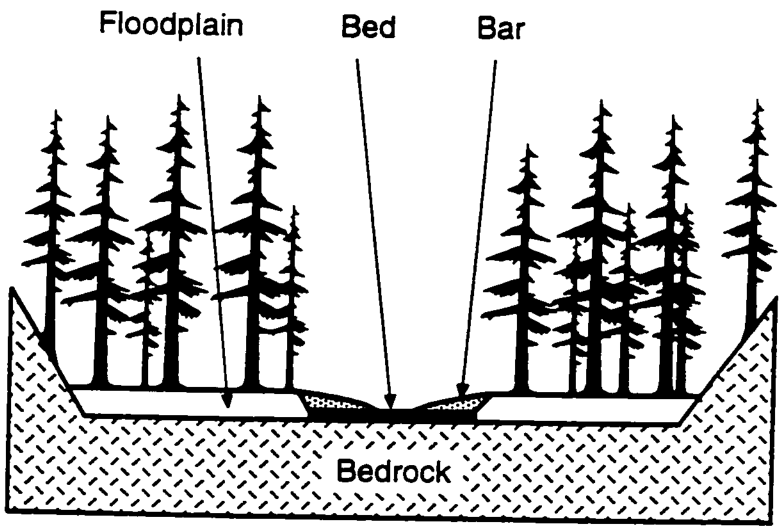


Figure 4-3. Schematic diagram defining locations of sediment storage features. Sediment stored in the channel bed (black area) directly overlies bedrock or quaternary sedimentary deposits. Bars (gray areas) are storage features within the bankfull channel and overlying the bed. The floodplain (white area) has a vegetation age greater than five years and is flooded less than annually on average.

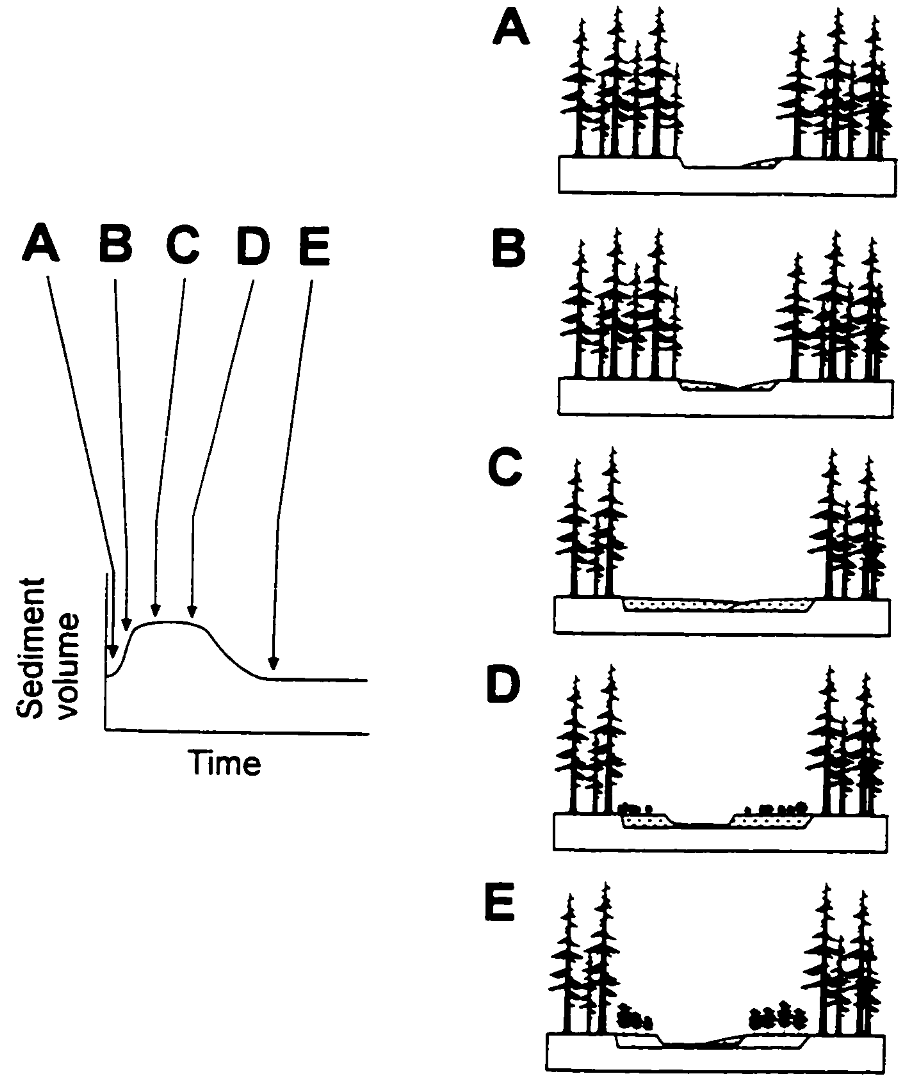


Figure 4-4. Time sequence of a channel cross section as a bed load wave passes through. (A) Prior to the arrival of a bed load wave, the channel has relatively low amounts of sediment stored in the bars and bed. (B) As a bed load wave begins to enter the reach, storage in the bed and bars increases, while floodplain storage does not change. (C) When a wave is fully in the reach, floodplain storage decreases as the channel widens, and bed and bar storage increase dramatically. (D) Within the first year, bed storage decreases but bar storage remains high. After the first year, the channel may tend to migrate laterally as long as the bed load wave resides in the reach. Shifting locations of bars inhibits revegetation. (E) Once the wave passes through the reach, sediment stored in large bars created by channel widening is transferred to flood plain storage as vegetation age increases. (Bars are reclassified as floodplain when vegetation age exceeds 5 years.)

Ozaki (1996). Prior to an increase in sediment supply, the channel has a relatively constant width and depth, and a clearly defined channel with relatively small bars (Figure 4-4a). As the wave enters the reach, the bed surface becomes finer, bars become larger, and pools may begin to fill (Figure 4-4b). As the main body of the wave enters the reach, sediment supply increases well beyond transport capacity and the channel bed aggrades and widens. At extremely high sediment supply the entire bed aggrades to flood plain level, and bed load sediment may be deposited on the flood plain (Figure 4-4c).

Recovery from high sediment supply appears to occur in stages with rapid recovery to approximately pre-aggradation channel geometry, but slower recovery of lateral stability and revegetation of the newly formed flood plain. Recovery of a defined channel within the aggraded bed usually occurs within a few months to a few years (Figure 4-4d). This newly formed channel often approximates pre-aggradation channel geometry even on the surface of a sediment wave. However, the channel is typically laterally unstable, and the thalweg location may shift frequently within the aggraded channel.

Recovery of the original channel width (i.e., revegetation of the flood plain) is dependent first on passage of the wave, and subsequently on lateral stability of the channel. It is not clear whether revegetation stabilizes the stream channel, or whether lateral stability of the channel allows revegetation to occur. Collins et al. (1994) found that the occurrence of large floods appears to destabilize recently revegetated channels, indicating that revegetation cannot by itself control the lateral movements of a channel. In the absence of unusually large floods, vegetation is re-established on most of the newly formed flood plain within a few years (Figure 4-4e). Stored sediment is here considered to be in bars until vegetation age reaches five years. At vegetation ages greater than five years, the storage reservoir is considered a flood plain.

Magnitude of effects of altered sediment supply on channel morphology

This section first provides background on the effects of changing sediment supply on channel geometry and morphology based on review of the literature. It then summarizes quantitative relationships between sediment supply and habitat characteristics described by Nelson (1998), which can be used to estimate the magnitude of sediment supply disturbance with respect to changes in habitat characteristics. The background section first describes the mechanisms by which changing sediment supply alters bed surface texture, channel geometry, and pool formation. Changes in bed surface texture are described first because they are the first response to a change in sediment supply. Changes in channel geometry are discussed next because they are theoretically only manifest when changes in bed surface texture are no longer capable of accommodating a change in sediment supply. Because pool depth is related both to sediment transport and channel geometry, it is discussed last. The second part of this section describes estimation of the magnitude of changes in habitat characteristics caused by a change in sediment supply. It is based primarily on Nelson (In preparation), which focuses on reach-level effects of altered sediment supply on habitat characteristics.

Background

Most research into channel responses to changing sediment supply has been conducted in low-gradient depositional areas, with very little attention paid to steeper channels. From the standpoint of anadromous salmonid resources and the restoration approach presented in Chapters 2 and 3, that focus is appropriate since those reaches are precisely where the majority of anadromous salmonid habitats lie. In general, increased sediment supply in low-gradient reaches (slope < 0.02) can cause fining of the bed surface (Dietrich et al. 1989), severe channel widening (Kelsey 1982b, Madej 1982), channel aggradation (Madej 1982, Lisle 1982), and reduction of pool depth (Lisle 1982, Collins et al. 1994, Madej and Ozaki 1996). Channels may also become laterally unstable (Bergstrom 1982, Church 1983), which may hinder revegetation of unvegetated bars and floodplains (Collins et al. 1994). This section describes three common effects of changes in sediment supply: fining of the bed surface, altered channel geometry, and pool shallowing, focusing on the mechanisms by which such changes occur.

One effect of increased sediment supply is fining of the surface layer of the bed material (e.g., Dietrich et al. 1989). Dietrich et al. (1989) proposed that coarsening of the bed surface relative to the subsurface results from an imbalance of sediment transport to sediment supply for stream beds composed of many different grain sizes. They proposed a dimensionless sediment transport ratio q^*, which is the sediment transport rate of the surface normalized by the transport rate of a surface as fine as that of the bed load. Dietrich et al. (1989) expressed sediment transport rate per unit channel width (q_b) as function of excess shear stress

$$q_b = k(\tau_b - \tau_c)^n$$

and the dimensionless sediment transport ratio as

$$q* = \left(\frac{\tau_b - \tau_{cs}}{\tau_b - \tau_{ci}} \right)^{1.5},$$

where k and n are empirical constants, τ_b is basal shear stress, and τ_{cs} and τ_{cl} are the critical shear stresses of the surface and subsurface, respectively. They predicted that $q*$ should equal one when sediment transport equals sediment supply, and that $q*$ should decrease toward zero as the bed surface coarsens when supply is reduced. In other words, decreased supply of sediment leads to armoring of the bed surface, which increases the critical shear stress of the bed and decreases sediment transport rate. Flume studies conducted by Dietrich et al. (1989) and others supported the hypothesis. They interpret their results to mean for natural channels that increases in sediment supply can be accommodated through an increase in q_* to the point that q_* reaches a value of one for most of the bed surface (i.e., the bed surface is not armored), and that thereafter increases in sediment supply must be accommodated by a change in channel morphology.

Fining of the bed surface with increasing sediment supply has several implications for sediment transport and channel morphology. As the bed surface becomes finer, the threshold of bed movement decreases. Sediment transport then begins at lower basal shear stress, which is generally equivalent to shallower flows. Bagnold (1977), presents an equation expressing sediment transport rate (q_b) as a function of excess stream power:

$$q_b = \frac{1.6(\omega - \omega_c)\left(\frac{\omega - \omega_c}{\omega_c} \right)^{1/2}}{\left(\frac{\bar{h}}{\bar{D}_{sed}} \right)^{2/3}}$$

where ω is stream power per unit bed area ($\omega = \rho gqs$), ω_c is critical stream power, h is flow depth, D_{sed} represents the size of the bed load, ρ is the density of water, g is acceleration due to gravity, q is unit stream discharge, and s is water surface slope. Based on this equation, sediment transport rate is a function of sediment size in the parameter h/D_{sed}. Decreases in D_{sed} tend to reduce sediment transport rate. The parameter ω_c is also a function of sediment size, and decreased D_{sed} results in a decrease in ω_c and an increase sediment transport rate. Therefore, predicting the change in sediment transport rate at a given stream power is a function of the relationship between ω_c and D_{sed}.

For a channel of given slope, critical stream power per unit bed area ($\omega_c = \rho g h_c s$) and critical shear stress ($\tau_c = \rho g q_c s$) are primarily functions of the sediment characteristics including size (D_{sed}), drag coefficient (C_D), angle of repose of sediment (θ) and specific gravity (γ_s). Each of these are components of the forces that resist movement of the bed load sediment. Assuming that C_D, θ, and γ_s of individual grains change little with increasing sediment supply, dimensionless critical shear stress (τ_{*c}) can be expressed as

$$\tau_{*c} = \frac{\tau_c}{(\gamma_s - \gamma)D_{sed}} = f\left(\frac{U_{*c} D_{sed}}{\mu / \rho}\right)$$

(ASCE 1977, equation 2.108), where $U_{*c} = \sqrt{\tau_c / \rho}$, μ = viscosity, and $\gamma = g\rho$. If τ_{*c} is essentially a constant for natural gravel channels (see discussion in Buffington 1995), then fining of the bed surface with increasing sediment supply indicates that τ_c decreases up to the point at which fining of the bed surface can no longer accommodate the increase in sediment supply (i.e., the point at which sediment supply = sediment transport). When sediment supply exceeds sediment transport, deposition must occur and the channel aggrades. Furthermore, during aggradation and until sediment supply diminishes to below sediment transport, the bed surface theoretically retains a size distribution equal to that of the subsurface (i.e., $q^* = 1$), indicating that sediment transport is at maximum efficiency. Therefore, in aggraded channels it is reasonable to assume that the bed surface is finer than in the non-aggraded case, and that τ_c is at a minimum for that channel and grain size distribution.

When fining of the bed surface no longer accommodates changes in sediment supply, the channel responds by changes in geometry (Dietrich et al. 1989). Although such changes theoretically only occur when most of the channel bed has a surface grain size approximating that of the subsurface, it is likely that localized aggradation begins on parts of the bed where $q_* = 1$, even when much of the bed remains armored. Such localized changes might be observed at low to moderate sediment supplies. However, at high sediment supplies (i.e., where sediment supply dramatically exceeds transport capacity) most of the bed surface will be unarmored and the channel response will approximate that hypothesized by Dietrich et al. (1989). In these cases, aggradation of the channel bed and channel widening are typical (e.g., Lisle 1982, Madej 1982, Madej 1992, Harvey 1987, Harvey 1991).

To provide a context for field observations of local channel widening, Andrews (1982) summarized a theory of near constant stream channel width over time for self-formed stream channels. In this quasi-equilibrium context of channel form, Andrews noted that aggradation should be concentrated near the stream banks (Einstein 1972 as cited in Andrews 1982), and that channel widths should therefore tend to narrow when sediment is deposited. Such deposition is more likely when channels are too wide for given flow and sediment conditions, leading to narrowing of the channel (Parker 1978 as cited in Andrews 1982). By contrast, when a channel is too narrow for given conditions, shear stresses at the bank are larger, and the channel tends to widen. Andrews' (1982) field observations in the East Fork River, Wyoming generally supported these predictions. Cross sections that were narrower than the average tended to scour near the banks at discharges greater than bankfull. The banks consequently became undercut and unstable, and the channel widened. Conversely, channel width was unchanged at wider cross sections where deposition occurred at discharges greater than bankfull.

While apparently adequate for relatively steady supplies of sediment, Andrews' theory does not explain numerous observations of dramatic channel widening and shallowing in response to extreme increases in sediment supply (Lisle 1982, Madej 1982, Madej 1992, Harvey 1987, Harvey 1991). Andrews' theory predicts narrowing of a channel when sediment supply increases because deposition occurs near the banks. However, numerous observations of channel widening under high sediment supply contradict the theory, even though the mechanisms driving such widening are poorly understood. Bagnold (1977) suggested that the widening response may be a function of increased "protection" of the stream bed against downcutting and consequent erosion of the stream banks. In other words, as the stream bed aggrades due to an increase in sediment supply, an increasingly thick layer of sediment on the bed reduces erosion of the bed and shallows the flow. Shear stress is therefore relatively higher against the banks, and the banks must erode. Flow may also increase across the flood plain, which increases the potential for erosion of banks or portions of the flood plain surface.

Subsequent to aggradation and widening, channels tend to incise and narrow to pre-aggradation widths or less as sediment supply is reduced (Harvey 1987, Pitlick and Thorne 1987). The timing of this narrowing depends on the rate of reduction in sediment supply and the incidence of storms and floods after the initial aggradation (Harvey 1987). Harvey (1987) found that some aggraded channels in England had incised and narrowed to less than pre-aggradation widths within four years, and Pitlick and Thorne (1987) found that the upper 600 m of a 2000 m aggraded reach in a Colorado stream narrowed to pre-aggradation width within one year. Both indicate that recovery to pre-aggradation channel geometry can occur within a few years under conditions of reduced sediment supply.

As sediment is exported from an over-supplied reach, more and more of the bed surface should become armored. Armoring in such cases appears to be due primarily to winnowing of fines from the surfaces of high shear stress locations in the channel as flood flows recede (e.g., Beschta 1987). Such armoring would be expected to occur first along the thalweg in riffles, and then to expand gradually into pools and finally onto bar surfaces. As the armor layer develops across the bed, τ_c will gradually increase leading to a decline in sediment transport.

Local observations indicate that incision into an aggraded channel bed can occur during the first moderate floods after aggradation, and that formation of a post-aggradation channel with dimensions only slightly smaller than the pre-aggradation channel may occur in much less than one year depending on the timing of floods (J. Thompson, Geologist, Lummi Tribe, Bellingham, Washington). This observation is supported by the fact that shallow and wide aggraded channels without incision were extremely rare in the Skagit study area, whereas aggraded and widened channels with a smaller channel incised into recent deposits were quite common. In most cases, the surfaces of recent deposits were unvegetated or sparsely vegetated, and the elevation of the surface was typically at or near the elevation of the flood plain. Hence, recovery of channel geometry appears to occur within months or years if sediment supply is reduced. However, location of the recently incised channel appears to be laterally unstable compared to that of a pre-aggradation channel (Harvey 1987).

Increased sediment supply appears to reduce depths of pools independently of pool-forming mechanisms (Lisle 1982, Collins et al. 1994). In some cases pool depths are not restored until the "wave" of sediment passes through the reach (Lisle 1982, Collins et al. 1994). However, there are some indications that pool depths may begin to recover even before the wave of sediment has passed through a reach (Madej and Ozaki 1996), although pools remain shallower in an aggrading reach than in degrading reaches. In the Trinity and Eel River basins of northern California, Lisle (1982) noted a roughly two-fold increase in the ratio of residual pool depth to total depth in two channels that had degraded after aggradation resulting from a large storm in 1964. Degradation of the two sites occurred over a period of less than 15 years. Collins et al.

(1994) found that average residual pool depth over a 13 km reach of Deer Creek in northwest Washington had tripled within 6 years after the initial aggradation caused by a single large landslide in the basin. Madej and Ozaki (1996) also found that pool depths tripled over a period of six years after extensive mass wasting in Redwood Creek in northern California. Such observations indicate that the recovery of pool depths after aggradation can occur within a few years, a time-frame not much different from that of the recovery of channel geometry.

Despite documentation of pool shallowing resulting from channel aggradation, we have little understanding of the mechanism by which pools become shallower. In the case of a large increase in fine sediment supply to a gravel bed channel, pool filling can be explained by transport of fine sediments from riffles and deposition in lower velocity pools at low discharges. This explanation relies on the notion of size-selective transport, which can be simplified to a two-phase pattern of bed load transport (Beschta 1987). Phase I transport refers to the transport of sand-sized particles formerly deposited in low velocity areas such as pools. During Phase I transport, stream discharge (and consequently shear stress on the bed) is sufficient to mobilize smaller particles, but not sufficient to disrupt the gravel-sized armor layer. As discharge increases, the armor layer is disrupted resulting in Phase II transport, or transport of coarse bed load. Implicit in this model of bed load transport is that cessation of Phase II transport occurs prior to cessation of Phase I transport as stream discharge recedes. Furthermore, as stream discharge continues to decline, Phase I transport ceases in a spatially discontinuous fashion. Higher shear stress on riffle beds can continue to transport finer sediment while the lower shear stress in pools causes deposition. Hence, pools are partially filled by fine sediment scoured from riffles upstream.

An increase in coarse or mixed bed load cannot be expected to shallow pools by the same process. To some degree, an increase in a mixed-size sediment supply may increase the magnitude of Phase I transport relative to that of Phase II transport due to the fining of the bed surface, resulting in some increase of fine sediment deposition in pools. However, the more likely effect is a decrease in the threshold of Phase II transport, and an overall increase in the magnitude of Phase II transport. Hence, the model of size-selective transport becomes less relevant to coarse bed load movement and the filling of pools.

Lisle (1982) proposed two potential mechanisms for pool shallowing due to an increase in the supply of coarse sediment. First, he noted that fining of the bed surface results in a decreased threshold of bed load movement. Noting the suggestion of Parker and Peterson (1980) that gravel bars are formed only at flood stages because sufficient sediment transport does not occur at lower flows, Lisle interpreted a decreased threshold of motion to mean that morphological features would be formed at lower flows. Lisle (1982) further stated that it is reasonable to expect bar amplitude to be smaller when formed at lower stages. With lower overall bed relief, pools would also be shallower because riffle crests (bars) between pools would be lower. Furthermore, erosion of riffle surfaces as discharge decreases would result in additional sediment deposited in pools, and would reduce the height of the control section (i.e., the riffle crest at the tail of a pool). Both effects would reduce overall bed relief and residual pool depth.

Second, Lisle (1982) noted that pool depth increases with increasing stage up to the stage at which sediment transport begins, then decreases, but increases again with increasing velocity (based on scour around bridge piers measured by Jain and Fischer 1980). He infers from this that increased sediment transport at lower flows would tend to decrease pool depths because equilibrium scour depth occurs when hydraulic scour is balanced by sediment infilling (Jain and Fischer 1980), and such an equilibrium would be reached at lower discharge when the threshold of bed load movement is lower. These effects can be illustrated with a plot of scour depth as a function of flow velocity normalized by critical flow velocity (U/U_c) (Figure 4-5). When sediment supply increases from below the rate of sediment transport to greater than the rate of sediment transport, both the surface grain size (D_{sed}) and flow depth (h) will decrease. Because increased sediment supply decreases the relative armoring of the bed and τ_c decreases, the bed load will be mobilized at lower discharges. Hence, one expects the threshold of bed movement to be reached at lower velocity, and that pool depths at critical velocity would be shallower in an aggraded channel than in a non-aggraded channel with an armored bed. In addition, total available τ will decrease in an aggraded channel that is shallower and wider than in the non aggraded case, because the flow depth (or discharge per unit width) will decrease. Both factors (lower critical shear stress and lower total shear stress) will tend to decrease pool depth.

To summarize, there are three main responses of a channel to increased sediment supply. As sediment supply to a channel begins to increase, the channel first responds by fining of the bed surface. This response tends to increase sediment transport capacity by reducing critical shear stress. When fining of the bed surface no longer accommodates increased sediment supply, the

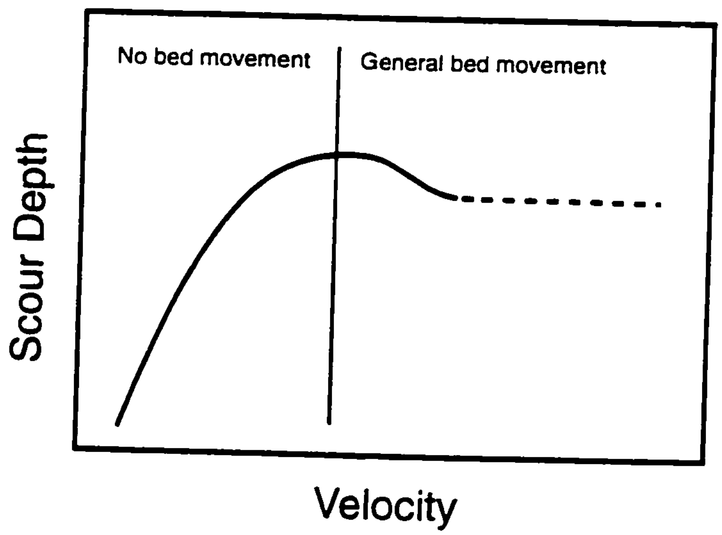


Figure 4-5. Schematic diagram of changes in scour depth at a bridge pier as a function of flow velocity normalized by critical flow velocity (U/U_c) (adapted from Jain and Fischer 1980). Scour depth increases with increasing velocity until the bed is in motion, at which point sediment transport into the pool begins to exceed the rate at which sediment is scoured from the pool.

channel then responds by bed aggradation and channel widening. This morphological adjustment is common in stream channels with significant increases in sediment supply. Shallowing of pools can occur along with fining of the bed surface because sediment transport and bar formation occur at lower discharges and bar amplitude and pool depth reflect the flow depths at which they are formed. Pool shallowing also occurs with channel aggradation and increased channel width for the same reasons.

Quantifying effects of changing sediment supply on channel morphology and pool characteristics

Although the preceding discussion illustrates some of the theoretical elements of channel responses to increased sediment supply, there is little in the way of quantification of responses. Relatively few studies have tried to assess changes in morphology of natural stream channels caused by changes in sediment supply, and those that have made the attempt have focused on channel geometry and pools (e.g., Kelsey 1982, Lisle 1982, Collins et al. 1994, Madej and Ozaki 1996). Each study has been site-specific, and has provided no general relationship that can be extrapolated to other locations. Therefore, a second component of this project (Nelson 1998) attempts to systematically survey and quantify inter-relationships between changes in sediment supply, changes in LWD abundance, and channel morphology.

Nelson (1998) found that the most difficult aspect of developing relationships between sediment supply to a stream channel and residual pool depths was finding a suitable in-channel indicator of high sediment supply. After testing three prospective indices of sediment supply (q* [Dietrich et al. 1989], width-to-depth ratio, and a ratio of the volume of channel-stored sediment to volume of the channel), Nelson (1998) found that no single variable appeared to be a good indicator of sediment supply. Moreover, none of these variables appeared to explain changes in channel morphology or changes in residual pool depth. Therefore, Nelson (1998) used the average annual sediment supply to stream channels upstream of the sample reach as the sole index of sediment supply. Sediment supply values were based on the sediment budgets of Paulson (1997), and included only mass wasting sediment supply during approximately the last 20 years of the sediment budget. Sediment supply from soil creep and surface erosion was ignored because the variability in supply among basins was small compared to the variation in mass wasting rates.

The main drawback of this approach is that routing of sediment cannot be accounted for, and a high average sediment supply over a 20-year period may not mean that sediment supply is high in any given reach at the time of the survey. When the recent sediment supply is dominated by large events in one or two years, sediment delivered to the channel may not yet have reached the surveyed stream reach, or may have already passed through the reach. Thus, residual depths may not reflect a high sediment supply even though the sediment budget indicates a high average supply. Conversely, when a low average sediment supply includes a single mass wasting event near a surveyed reach, the local sediment supply may exceed the transport capacity of the reach. In this case, residual depths may reflect a high local sediment supply even though the averaged sediment budget does not.

Despite the potential errors induced by the approach, Nelson (1998) found a general relationship between sediment supply and residual pool depths. The average residual depth of pools increased as basin area increased, but pools tended to be deeper for a given basin area when mass wasting sediment supply was low (<90 $m^3/km^2/yr$, Figure 4-6). Pool depths at high sediment supply (<90 $m^3/km^2/yr$) are not below the range of depths for low sediment supply, but tend to be nearer the low end of the range. In other words, between-reach variability in residual pool depths is greater at low sediment supply than at high sediment supply, but the shallowest depths are approximately the same for both high and low sediment supply.

A regression of pool depth on basin area for sites of high sediment supply yields the equation

$$d_{res} = 0.0093(A) + 0.25,$$

where d_{res} is average residual pool depth in meters and A is basin area in km^2. Pool depths at low sediment supply have roughly equal probability of being above or below the regression line, whereas pool depths at high sediment supply are nearly always below the regression line.

Duration of effects of altered sediment supply on channel morphology

This study seeks to estimate recovery time from increased sediment supply for a large number of reaches across a landscape, which requires a method of estimating sediment transport rate that balances lower field effort against greater accuracy. Unfortunately, estimates of sediment transport by most sediment transport formulae are notoriously inaccurate when employed across a broad range of channel slopes or sizes, and even sophisticated models

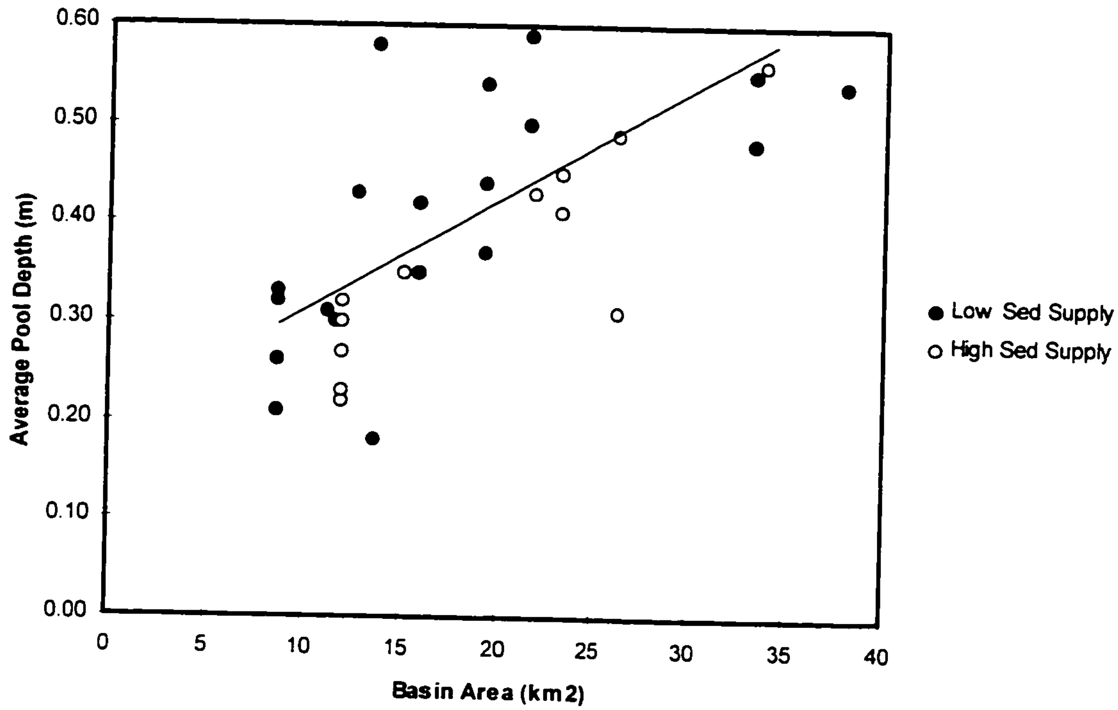


Figure 4-6. Average residual depth of pools plotted against basin area for sites with high sediment supply (>90 $m^3/km^2/yr$) and low sediment supply (<90 $m^3/km^2/yr$). Regression line is for low supply sites. Equation is $d_{res} = 0.0093(A) + 0.25$, n = 19, $r^2 = 0.43$. Note that the line approximates the upper limit of average pool depth for high sediment supply sites.

requiring extensive field effort and calibration typically produce poor results (Gomez and Church 1989). Therefore, I sought a simpler index of sediment transport rate that required little field effort, and predicted average annual travel distance rather than mass or volume transfer. This section first reviews a number of studies of sediment transport, paying particular attention to the accuracy of various predictive methods and to the limitations of various methods. This review provided a number of insights into why most sediment transport formulae are unable to represent sediment transport accurately, which helped guide the selection of parameters for indexing sediment transport rate.

Background

A major element of this dissertation is estimating the rate at which sediment moves through stream channels, especially under conditions of very high sediment supply. Three traditional approaches to estimating the rate of sediment movement through channels are (1) physically-based sediment transport formulae, (2) kinematic wave transport models, and (3) mass balance calculations. Physically-based sediment transport formulae are the most common, and have been used extensively for estimating rates of sediment movement through channels and for estimating aggradation or degradation rates. These models employ a variety of parameters that can be theoretically linked to sediment transport, such as slope, discharge, and particle size. Kinematic wave models typically employ some of the same physical parameters as the sediment transport formulae, but they also account for the effect of particle concentration on the rate of sediment movement. These models have been used less extensively, but they have been used successfully to estimate the rate of movement of pulses of sediment such as mining waste. The mass balance approach can be used to estimate sediment transport in a channel network if sediment inputs and changes in channel-stored sediment can be estimated. The rate of sediment transport is then input minus change in storage for each reach. This approach often incorporates physically-based formulae or other measures of sediment movement in channels and is often used as a component of modeling sediment transport or tracking the movement of sediment through a channel network.

A wide variety of physically-based formulae exist for estimating sediment transport in rivers (for examples see summaries in ASCE 1977, Richards 1982, Gomez and Church 1989). Gomez and Church (1989) grouped many of these formulae into four general approaches based on shear stress, stream discharge, stream power, or stochastic functions of particle movement. All such formulae use as inputs some combination of energy gradient (such as water surface slope), size of bed load particles, and a parameter representing water flow (usually depth or discharge). Gomez and Church (1989) noted that all equations are therefore essentially scale equations of the flow, and they did not find it surprising that no formula adequately described the physics of sediment transport. They also observed that many formulae were developed or tested with very little field data, and that there were more transport formulae than there were data sets with which to test them. They conclude from their assessment that no formula is capable of adequately predicting bed load transport in gravel bed rivers, especially in the case that no sediment transport information is available for the river. Nevertheless, they advocated use of the Bagnold (1980) formula for estimating the magnitude of bed load transport when only limited hydraulic information is available. This conclusion was based mainly on the appearance that stream power formulae provide the most straight forward scale correlation.

Most formulae for gravel-bed rivers have been developed with data from channels or flumes having slope < 0.01, which renders them inappropriate for our study area where many channel slopes are steeper than 0.04. However, other studies in western Washington have made use of bed load transport formulae, either because the site characteristics were similar to those used in developing or testing the formula (e.g., Collins and Dunne 1989) or because other research supported the approach (e.g., Booth et al. 1991). Collins and Dunne (1989) employed the bed load transport formulae of Parker et al. (1982) and Meyer-Peter (referencing its discussion by Raudkivi 1976) for rivers draining the southwest Olympic mountains. Following accepted practice for the Meyer-Peter formula, they restricted its use to reaches without large scale roughness elements such as large gravel bars or bends. They restricted their use of the Parker formula to channels with slopes <0.01 in keeping with range of flow conditions from which the relation was derived. In selecting an appropriate formula for estimating bed load transport in the Snoqualmie River, Booth et al. (1991) followed the suggestion of Gomez and Church and employed the Bagnold (1980) formula. Gomez and Church (1989) showed that predicted bed load transport by this formula is typically within a factor of two of measured values, but also that predictions were sometimes more than an order of magnitude different from measured values.

In some cases bed load transport formulae have been applied to steeper channels with moderate success. For example, Bathurst et al. (1987) used the formula described by Schoklitsch in 1962 (modified to allow for non-uniform particle size distribution) to predict both initiation of motion and bed load discharge. They suggested that the formula was valid for slopes ranging from 0.0025 to 0.10, but cautioned that the equation should not be applied to the entire bed when slope exceeds 0.01. In such cases, the equation should be applied separately to each particle size class. However, from their Figure 15.8 it is apparent that the formula overpredicts sediment discharge by 1-2 orders of magnitude in small streams.

Other researchers provide some insight as to why such formulae are unreliable for general application to gravel bed streams and rivers. Church and Hassan (1992) and Hassan and Church (1992) found that particle transport distance is a function of particle size, but also that the effect of particle size is reduced by the effect of hiding or burial of particles. In general, particle size strongly affected transport distance when scaled particle size (diameter of a particle divided by the D_{50} of the subsurface particle size distribution, D/D_{50sub}) is >2, with transport distance dropping to zero as D/D_{50sub} increased from 2 to about 10. When scaled particle size was < 2, particles had a higher probability of being trapped and their movement was relatively insensitive to particle size. Travel distances of particles were only weakly correlated with excess stream power, probably due at least in part to the effects of variable bed structure (Hassan et al. 1992). Furthermore, when mean particle travel distance approached the length of bar spacing, trapping of particles in bars appeared to reduce the rate of transport (Hassan et al. 1991).

A second factor is that the threshold of initial motion can be much higher than that of final motion, resulting in continued bed load transport as flow drops well below that at which transport began (Reid et al. 1985). This phenomenon is illustrated in plots of bed load sediment discharge against water discharge, and is apparently caused at least in part by armoring of the bed surface (e.g., Sidle 1988). Sidle (1988) noted that hysteresis is not as common or pronounced for bed load as it is for suspended load, probably because several other unpredictable factors also affect bed load discharge during transporting events. Depletion of available sediment and storage or release of sediment behind woody debris are also likely influences on sediment transport rate.

Sidle (1988) also noted that sediment discharge on the rising limb of the hydrograph sometimes exceeded that on the falling limb, but that on other occasions the reverse was true. Furthermore, he found opposite patterns at two bed load sampling stations approximately 20 m apart, suggesting that bed load moves in waves or pulses, and that the amount of sediment available for transport varies along the length of a channel. Meade (1985) drew a similar conclusion in describing bed load "waves" on the East Fork River, Wyoming. However, in addition to longitudinal variations in sediment availability, Meade (1985) also described the variation of bed load availability and transport across the channel width. With respect to bed load transport formulae, Ergenzinger (1988) noted that waves appeared to result from adjustments of the channel bottom, and that bed load functions only apply when the entire bed is in motion.

Bergstrom (1982) and Reid et al. (1985) inferred the movement of sediment in wave-like forms based on short term variations in sediment discharge data, and Meade (1985) noted that waves appeared to be groups of particles that moved from one low flow storage site to another. At a much larger scale, Madej and Ozaki (1996) observed a large wave of sediment in Redwood Creek, California that resulted from severe mass wasting during the large storm of 1964. The wave was still present and moving through the channel as of 1992, though much attenuated from its earlier amplitude (Madej 1992). Observed waves range in size from tens of centimeters high and hundreds of meters long (e.g., Meade 1985), to about 1 m high and several kilometers long (e.g., Madej and Ozaki 1996). Annual travel distance of bed load particles or waves range from hundreds to thousands of meters per year (e.g., Perkins 1989, Madej 1992). Most physically-based sediment transport formulae do not account for changes in sediment supply related to the passage of bed load waves, which may also contribute to the inaccuracy of those formulae (Gomez and Church 1989).

Despite numerous studies noting that aggradational zones tend to migrate downstream as a wave (e.g., Kelsey 1982b, Pickup et al. 1983, Meade 1985, Madej 1992), and that channel adjustments are related to the downstream migration of changes in the distribution of channel-stored sediment (e.g. Kelsey 1982b, Harvey 1987, Madej 1992, Madej and Ozaki 1996) there have been relatively few attempts to explain or model the movement of sediment pulses. In one early discussion of wave-like movement, Leopold et al. (1964) noted that only Bagnold had considered the concentration of grains as a factor in sediment movement until that time. Leopold et al. (1964) conceptually described the movement of sediment as kinematic waves in which individual particles move faster than the wave form. They suggested that two factors determine the velocity of a wave form: the transport rate (number of particles passing through a cross section per unit time), and the linear concentration of particles (number of particles per unit distance). Thus, transport rate is the product of mean particle velocity and particle concentration. They considered sand dunes and bars to be kinematic waves that tend to move downstream, and gravel bars to be typically stationary bed forms. However, they did not discuss the wave-like movement of coarse sediment.

Weir (1983) demonstrated that an equation for bed waves and their velocity can be developed from the assumption that water depth and velocity adjust instantaneously to changes in bed profile. The resultant equation could then be used to calculate bed wave velocity as long as sediment discharge was known. Weir then applied these equations to the East Fork River, Wyoming. Bed load transport was first calculated using two "empirical" sediment transport formulae, yielding predictions that were in general agreement with field measurements (within a factor of 2). The estimated bed load discharge rates were then used to estimate bed wave velocities, which were also in general agreement with field data. Although the approach did not directly model sediment transport as a moving wave, it did show that some characteristics of wave-like motion could be mathematically described and that model results were reasonable.

In modeling the transport of slugs of mining waste in the Kawerong River in Papua New Guinea, Pickup et al. (1983) treated sediment movement in a manner that was more true to principles of kinematic waves. Their dispersion model explicitly accounted for particle concentration, mean particle velocity, and the distribution of sediment velocity about the mean (represented by a dispersion coefficient). For comparison, they also modeled sediment transport using a continuity (mass balance) model which ignored variation in particle velocity. They found that both the dispersion and continuity models predicted a total volume of transported sediment that was of the correct order of magnitude, even though poor quality input data reduced the accuracy of both models. However, the dispersion model predicted sediment movement and deposition more accurately than the continuity model. Both of the wave models require extensive field data in order to apply them with reasonable accuracy.

The mass balance approach is essentially the foundation of the sediment budget (Dietrich et al. 1982). It describes transfer of sediment into and out of a reach, with inequalities between input and output equaling the change in stored sediment. In its simplest form, it is described by the equation:

$$I + \Delta S = O$$

where I is input; ΔS is the change in volume of stored sediment; and O is output. At least two parameters must be estimated independently, and the third can be estimated by difference.

Variations of the mass balance model have been used for a variety of purposes ranging from estimating residence time of sediment in different storage reservoirs of a large watershed (Kelsey 1982b, Madej 1992) to simulating sediment routing and storage in a channel network over several thousand years (Benda 1994). However, in many cases the bed load discharge or output (O) is calculated using one of the physical transport formulae, and the sediment budget's primary purpose is to estimate aggradation or degradation (e.g., Dietrich and Dunne 1978, Pickup et al. 1983, Booth et al. 1991). It appears relatively rare that the sediment budget approach has been used to estimate sediment transport by difference as was done by Roberts and Church (1986). Roberts and Church (1986) used a sediment budget to calculate that sediment transport rates in severely disturbed watersheds of the Queen Charlotte Islands had increased by up to 10 times, and that residence times of in-channel sediment had increased by as much as 100 times. This methodology also requires extensive field effort to document sediment storage, and was therefore considered unsuitable for application across a large landscape.

Based on the preceding literature review, no existing model is well suited to the task of rapidly estimating sediment transport rates from increased sediment supply across many stream reaches on a landscape. Few methods would appear to accurately predict recovery rates from increased sediment supply despite significant field effort required for calibration. Those that may be sufficiently accurate appear to require too much field effort to have practical application at the landscape scale. Although no existing method was considered suitable for this dissertation, all of the physically-based sediment transport equations indicate that energy slope, flow (discharge or depth), and particle size influence sediment transport capacity. Furthermore, there is significant evidence that large bed load pulses migrate through channels as waves, and that such waves have characteristic velocities that are at least grossly related to stream size. Finally, there is evidence that the annual travel distance of particles or waves are to some extent controlled by bar spacing. These insights will guide the development of simple indices that can be used to help evaluate recovery of channels from increased sediment supply across a landscape, which is described in the next section.

<u>Annual travel distance of bed load</u>

One of the major interests of this study is the rate of export of sediment from an aggraded reach. This export is in some way a function of the sediment transport capacity of the reach and the supply of sediment to the reach. It is clear that channel beds aggrade when sediment supply is significantly higher than the transport capacity of a reach, and that channel widening and reduced fish habitat quality often follow (e.g., Kelsey 1982b, Lisle 1982, Madej 1982, Harvey 1991, Collins et al. 1994, Madej and Ozaki 1996). If sediment supply to an aggraded reach drops below the sediment transport capacity of that reach, the channel will degrade until bed armoring inhibits channel incision (Bagnold 1977, Dietrich et al. 1989). At this point bed elevation appears to remain relatively constant over several decades (Madej and Ozaki 1996), and channel morphology and fish habitats return to pre-disturbance forms (Lisle 1982, Collins et al. 1994). This section focuses on the length of time that sediment resides in a reach, which is here considered the dominant factor affecting the duration of a sediment supply disturbance.

Annual travel distance is a function of the energy of a stream to transport sediment, and is strongly related to excess stream power or excess shear stress at the bed (Hassan and Church 1991, O'Connor and Harr 1994), duration of transporting events (Emmett et al. 1983) and grain size (Hassan and Church 1992, Church and Hassan 1992). One further expects that annual travel distance is affected by sediment supply to some degree. Exchange of individual grains with the bed appears to slow the rate of travel of the entire bed load (Hassan and Church 1991). However, in this study the main interest is estimating the rate of travel of sediment waves in reaches with high sediment supply. By focusing on rates of sediment movement under high sediment supply, I assume that differences in rates of movement due to differences in sediment supply will be small.

Several researchers have reported findings of annual travel distance along with drainage area or other channel characteristics that influence annual travel distance such as sediment size (Hassan and Church 1992, Church and Hassan 1992) and channel slope (Emmett et al. 1983, Kelsey 1982b). Some researchers have specifically compared annual travel distance to flow velocity. However, travel distance appears to be most generally related to stream power (e.g., ρgQs) or basal shear stress (ρghs), and the duration of bed load transporting flow. Within a hydrologic region, stream power and basal shear stress should be correlated with drainage area because Q and h are correlated with drainage area (e.g. Leopold et al. 1964). Furthermore, the cumulative duration of sediment transporting events should be relatively similar because streams within a hydrologic region should have similar amounts and types of precipitation. Annual travel distance at some specific discharge (e.g., a bankfull flow for example) should therefore be related to drainage area if hydrologic regime is similar among river basins.

In this section I develop a simple empirical approximation of sediment transport rate in order to estimate recovery time from an increase in sediment supply. As a first approximation, I assume that bed load annual travel distance (L_b) remains relatively constant as sediment supply changes, and that most of the change in bed load transport rate (as sediment supply increases or decreases) is in the depth of the mobile bed load layer, or in the length of the bed load wave. That is, as sediment supply increases or decreases there is little change in the average distance that particles move in a year, but significant changes in the apparent depth of the moving bed load layer, the length of the bed load sediment wave, or both.

To develop empirical relationships between bed load annual travel distance (L_b) and drainage basin area (A_d), bankfull channel width (w_{bf}), bankfull channel depth (h_{bf}) or bankfull discharge (Q_{bf}), I selected data from the literature where bed load annual travel distance was measured by tracer particles (e.g., Emmet et al. 1983) or by measurement of sediment wave movement (Table 4-2). Rates of wave movement were estimated by: (1) measuring movement of maximum channel widening on sequential aerial photographs (G. Pess unpublished data, WDNR 1994, Kelsey 1982b), (2) measuring movement of the scoured zone at the trailing edge of the wave (Pitlick and Thorn 1987), (3) measuring movement of the midpoint of channel-stored sediment (Madej 1992), or (4) combining mass-balance calculations with estimates of bed load discharge (Madej 1982, Perkins 1989). Because I am most interested in recovery from large increases in sediment supply, I sought annual travel distance measurements where the moving layer (or wave) is relatively thick and exchange of particles with the bed load is high. I also focused on channels with slope less than 0.03, which is where most anadromous salmonid habitats are located. With these restrictions, there were few data available with which to explore relationships between physical characteristics of channels and the rate at which bed load particles move through channels.

Hydrologic regimes vary considerably among sites described in Table 4-2 because stream locations range from Alaska to the Pacific Northwest to Wyoming. Unit discharge varies by more than an order of magnitude among sites (0.06 to 1.07 $m^3/s/km^2$), indicating that drainage area is not a suitable indicator of stream power across all reaches in the data set. It therefore did not appear reasonable to use drainage area (A_d) alone as one of the possible predictors of bed load annual travel distance. Bankfull discharge alone (Q_{bf}) and channel slope (s) alone were equally poor predictors of bed load annual travel distance (L_b). Linear regressions of L_b on Q_{bf} and L_b on s were not significant (p = 0.21 and 0.32, respectively). When combined in the quantity stream

Table 4-2. Summary of bed load annual travel distance and selected reach characteristics for channels with slope <0.03. Abbreviations are: Q_{bf} = bankfull discharge, w_{bf} = average width of bankfull channel, d_{bf} = average depth of bankfull channel, D_{50} = median grain size of bed load, L_b = annual travel distance of sediment, and A_d = drainage area. Letter codes after number of years indicates method of measurement or estimation: mw = measured movement of maximum channel widening, ms = movement of midpoint of channel stored sediment, t = tracer particles, cm = calculated based on measured sediment discharge, ce = calculated from estimated sediment discharge, fm = field measurement of post-wave bed degradation.

Area	Stream	Drainage Area (km²)	Q_{bf} (m³/s)	Slope	w_{bf} (m)	d_{bf} (m)	D_{50} (m)	L_b (m/yr)	Years	Reference
Slope ≤ 0.03										
N. Cascades	N.F. Stillaguamish	420	240[a]	.0029	56.2	1.6	gravel	1174	17 (mw)	Pess (unpub)
N. Cascades	Hansen	8	2	.022	11	~1	.025	104	27(mw)	WDNR (1994).
N. Cascades	Cornell	14	15[m]	.03	8		gravel	150		
N. California	upper Van Duzen	~80[k]	79[l]	.015	40[e]		gravel	890[d]	10 (mw)	Kelsey (1982b)
Puget Sound	Big Beef	38	18.9[f]	.01	20.5[g]		gravel	206	(ce)	Madej (1982)
W. Washington	Salmon	9.1	7.1[h]	~.02	9		.02-.06	240	3 (cm)	Perkins (1989)
N. California	Redwood	523	379[i]	.003	55			912	16 (ms)	Madej and Ozaki (1996)
Alaska	Toklat	267	37.5	.018	76.2[b]	.32[b]	.008	1650	(t)	Emmet et al. (1983)
Wyoming	East Fork River	206[c]	20	.0007	17	~1.2	.0013	650	(t)	Emmet et al. (unpub)
Colorado	Fall River	90	5[j]	.0015	11.5	0.75	gravel	600	1 (fm)	Pitlick and Thorne (1987)

a. Interpolated from base discharges per km² at USGS gages at Darrington (A_d = 213 km²) and Arlington (A_d = 679 km²)
b. Estimated from $w_{bkf} = 6.23 Q_{bkf}^{0.69}$ and $d_{bkf} = 0.225 Q_{bkf}^{0.95}$, Emmet et al. (1983)
c. Estimated from $A_d = \sqrt[0.68]{Q_{bkf}/36}$; rearranged from $Q_{bkf} = 36 A_d^{0.68}$, Dunne and Leopold (1978)
d. Movement of locus of maximum channel widening
e. Average of 26 m wide reach and 53 m wide reach
f. Estimated from regional curve in Dunne and Leopold (1978)
g. Estimated from Figure 2 in Madej (1982)
h. Estimated from $Q_{bkf} \sim A_d^{0.89}$ in Perkins (1989)
i. Discharge of 2-year recurrence interval flood
j. Based on base discharge from USGS gage at Estes Park (drainage area = 103 km²)
k. Each basin assumed to occupy ½ total drainage area, based on Figure 1, Kelsey (1982b)
l. Based on base discharge from USGS gage on Little Van Duzen River (drainage area = 94 km²)
m. Based on base discharge from USGS gage on Canyon Creek (drainage area = 22.5 km²)

Table 4-2 (cont.). Summary of bed load annual travel distance and selected reach characteristics for channels with slope > 0.03. Abbreviations are: Q_{bkf} = bankfull discharge, w_{bkf} = average width of bankfull channel, d_{bkf} = average depth of bankfull channel, D_{50} = median grain size of bed load, and A_d = drainage area. Letter codes after number of years indicates method of measurement or estimation: t = tracer particle measurements, ce = calculated from estimated sediment discharge.

Area	Stream	Drainage Area (km^2)	Q_{bkf} (m^3/s)	Slope	w_{bkf} (m)	d_{bkf} (m)	D_{50} (m)	L_b (m/yr)	Years	Reference
Slope > 0.03										
W. Oregon	Rock	16.6	24.4	.067				220	(ce)	Dietrich (1975)
Olympics	Eight-ten	.31		.16	3.3		.023	8[n]	4 (t)	O'Connor & Harr (1994)
Olympics	Sister	.53		.14	2.8		.014	41[n]	4 (t)	O'Connor & Harr (1994)
Olympics	Ramp	1.12		.14	4.1		.017	118[n]	4 (t)	O'Connor & Harr (1994)

n. Estimated average from graphs in O'Connor and Harr (1994) power per unit channel length ($\Omega = \rho g Q_b s$), slope and discharge are better predictors of annual bed load travel distance (Figure 4-7). However, the correlation remains poor when sites from all regions are included ($L_b = 355 + 72\Omega$, $r^2 = 0.35$, p =0.07).. Based on the available data, annual travel distance appears to be higher in streams with floods resulting from spring or summer snowmelt than in streams with rainfall or rain-on-snow floods. Furthermore, the Alaska stream is a braided, glacial meltwater channel which may have a sediment transport regime unlike any other in the data set. Excluding data from sites where sediment transport occurs primarily during snow melt floods, estimates of annual travel distance are about 200 m less ($L_b = 94 + 85\Omega$, $r^2 = 0.68$, p = 0.02). Finally, if we assume that L_b is zero when Ω is zero, then $L_b \cong 96\Omega$ ($r^2 = 0.66$, and p = 0.02).

Reasons for the apparent systematic difference in L_b between snowmelt-dominated and rainfall-dominated streams are unclear because several factors may be influencing L_b. Two of the three snowmelt sites (East Fork River in Wyoming and Toklat River in Alaska) had median bed load grain sizes in the sand or small gravel range (< 8 mm), whereas grain sizes at most of the rainfall sites were in the gravel range (2 to 64 mm). Initiation of motion of the finer sediments would tend to occur at lower discharges, possibly resulting in an increase in the cumulative duration of sediment transporting events during a year. However, the fact that grain sizes were not consistently finer in the snowmelt sites suggests that this is a partial explanation at best. A second possible factor is that snowmelt floods tend to be of longer duration than floods occurring during short intense rain storms, which may also cause an increased cumulative duration of sediment transporting events for spring snowmelt systems. In snowmelt systems, Emmet et al. (1983) noted that sediment transport occurred during 10 to 20 days each year in the East Fork River, Wyoming, and Parker and Peterson (1980) noted that the bed is active 30 days each year in Elbow River, Alberta. By contrast, sediment transport at rainfall dominated sites occurred only 5 to 6 days per year over a six year period in Southeast Alaska (Sidle 1988), and 11 days per year in Oak Creek, Oregon (Parker and Peterson 1980). Such differences in duration of sediment transport could account for higher annual transport distances in the snowmelt systems. Most large, sediment-transporting floods in the study area occur during fall-winter rain storms or rain storms combined with snow melt. Hence, it is reasonable to use the second relationship (excluding snowmelt systems) for a crude prediction of bed load annual travel distance in the

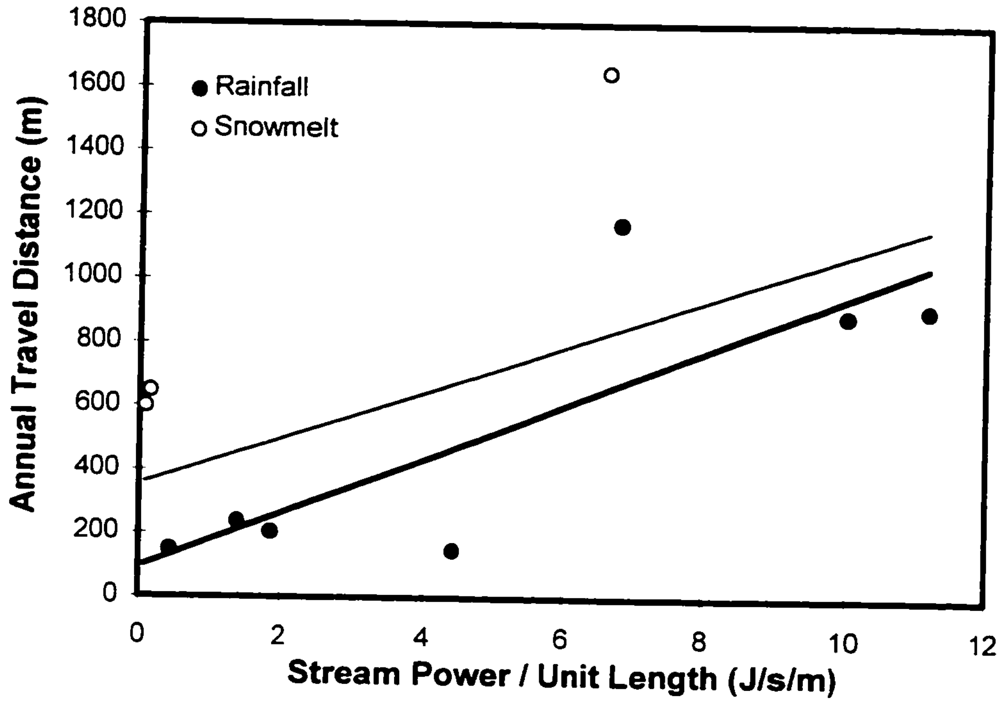


Figure 4-7. Relationship between bed load annual travel distance (L_b) and stream power per unit length (Ω). "Rainfall" data points are those for west coast streams where most floods occur during fall and winter rain storms. "Snowmelt" data points are those for the intermountain west and Alaska where floods typically occur during spring or summer snowmelt. For all sites combined, linear regression yields $L_b = 355 + 72\Omega$ (light line; $r^2 = 0.35$, $p = 0.07$). Excluding data from sites where sediment transport occurs primarily during snow melt floods, linear regression yields $L_b = 94 + 85\Omega$ (heavy line: $r^2 = 0.68$, $p = 0.02$).

study area.

It is also apparent from the preceding literature review that bar spacing is likely to influence the travel distance of bed load particles. That is, the movement of most particles on the bed is limited by trapping of particles in bars when travel distance approaches the average distance between bars (Hassan et al. 1991). Therefore, one expects that annual travel distance of bed load particles (L_b) should be related to bar spacing in some way. It is also well known that bar spacing is typically rhythmic in alluvial stream channels (e.g., Keller and Melhorn 1978). Bar spacing tends to occur at 5 to 7 channel widths measured along the channel in both meandering and straight channels (Leopold et al. 1964, Richards 1982), which is consistent with Yalin's (1971) theoretical prediction that large roller eddies (and associated bars) would form at a spacing of 2π channel widths. Thus, if L_b is a function of bar spacing and bar spacing is a function of channel width, L_b should also be a function of channel width.

Figure 4-8 shows the relationship between L_b and w_{bkf}, where w_{bkf} is bankfull channel width. Linear regression of all data yields $L_b = 84 + 19(w_{bkf})$, ($r^2 = 0.86$, $p = 0.0001$). Annual travel distance in the two intermountain snowmelt basins are 38 and 52 channel widths per year, which is approximately double the annual travel distances in coastal rainfall basins (14 to 26 channel widths per year). This could well be a result of the greater cumulative duration of sediment transporting events in the intermountain region (10 to 30 days in the intermountain region compared to 5 to 11 days in the coastal region). Excluding data from sites where sediment transport occurs primarily during snowmelt floods, linear regression yields $L_b = -34 + 20(w_{bkf})$, ($r^2 = 0.93$, $p = 0.0005$). The regression intercept is not significant, suggesting that the annual travel distance of bed load particles averages about 20 channel widths per year. That is, $L_b \cong 20(w_{bkf})$ in channels with slopes less than 0.03. This suggests that on average, bed load particles move a distance of one bar spacing every two to four transport events (i.e., an "average" particle moves about three bars per year during five to 11 transport days). This estimate of average annual travel distance of bed load sediment may also be useful in estimating recovery time from increased sediment supply.

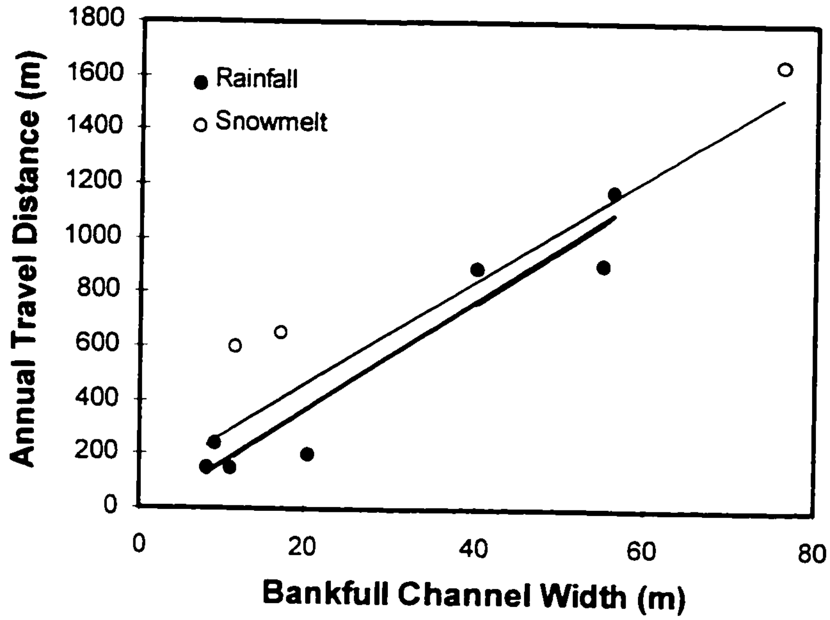


Figure 4-8. Relationship between bed load annual travel distance (L_b) and bankfull channel width (w_{bkf}). Linear regression of all data yields $L_b = 84 + 19w_{bkf}$, (light line: $r^2 = 0.86$, $p = 0.0001$). Excluding data from sites where sediment transport occurs primarily during snow melt floods, linear regression yields $L_b = -34 + 20w_{bkf}$, (heavy line: $r^2 = 0.93$, $p = 0.0005$).

Estimating recovery potential and recovery time

The recovery *potential* of stream reach is a measure of the "resilience" of a stream reach to an increase in sediment supply. A more resilient system has the capacity to recover more quickly after a disturbance. In the context of sediment supply disturbances, recovery potential should therefore be a measure of the rate at which a stream reach can either export sediment from an aggraded reach or transfer sediment into long term storage areas such as the floodplain. Here I assume that the primary recovery mechanism is transport of sediment through a reach, and that sediment transfer to floodplain storage is not a significant factor. Recovery potential for sediment supply disturbances is therefore defined as the average annual travel distance of sediment (in meters/year), which is the rate at which sediment waves move through a stream reach. Stream reaches which can transport sediment at higher rates are more resilient, and are considered to have greater recovery potential. Recovery *time*, by contrast, is dependent both on sediment transport rate and on the amount of sediment supplied to the channel. For sediment supply disturbances, recovery time is the estimated number of years required for a particular reach to recover from an increase in sediment supply, which is the amount of sediment to be exported divided by the transport rate. In general, lower recovery potential (i.e., slower sediment movement) combined with larger sediment supplies (i.e., more sediment to move) will result in the longest recovery times.

As discussed in the previous section, the empirically derived estimators of annual travel distance used here (i.e., $L_h = 92 + 85\Omega$, and $L_h \cong 20w_{bkf}$) are based on data from sites with high sediment supply and similar hydrologic regime. The first restriction (only including sites with high sediment supply) was intended to minimize the influences of large differences in sediment supply on the empirical relationships. That is, I excluded low supply sites where limited sediment availability is expected to increase annual travel distance (e.g., Leopold et al. 1964). The second restriction (including only sites with similar hydrologic regime) was intended to minimize the influence of variation in number of transporting events per year on annual travel distance. For this restriction I excluded snowmelt-dominated systems where the number of transporting events per year was roughly double that of the rain-dominated systems.

Recovery potentials of reaches in this study are mapped based on estimates of annual travel distance of sediments. The annual travel distance estimate is based on the regression of annual travel distance on stream power as shown in Figure 4-7. Stream power for individual reaches is estimated using stream slopes calculated from USGS 7.5' topographic maps, and estimates of flood discharge with a 2 year recurrence interval are calculated by regressions described in Cummans et al. (1974). The regressions are of the form $Q_2 = aA^bP^c$, where Q_2 is the discharge of a 2-year flood in cfs, A is drainage area in square miles, and P is annual precipitation in inches.

In general, steeper or larger streams have greater recovery potential (higher annual travel distances) and small, low-gradient streams have the lowest recovery potential (low annual travel distance), as illustrated for a five-square mile area of Bacon Creek sub-basin (Figure 4-9). Small tributaries such as Oakes Creek have very low annual travel distances in low-gradient reaches that traverse the terraces or floodplain of Bacon Creek, whereas larger tributaries such as Falls Creek have relatively higher annual travel distances across the floodplain due to greater discharge. (Oakes Creek and Falls Creek have drainage areas of about 1.2 km^2 and 13.5 km^2, respectively, and both streams have slopes of about 0.02 as measured from the USGS 7.5' topographic map.) However, large low-gradient streams have highest travel distances because discharges are 2 to 3 orders of magnitude higher than in small streams, whereas slopes of steeper streams are only 1 to 2 orders of magnitude greater than those of low-slope streams.

Because recovery potential is defined as the average rate at which a sediment moves through a given reach, recovery time for a specific location can be estimated readily if the size of the sediment wave is known. The most important measure of the sediment wave for this estimate is length (l). If length and annual travel distance (L_h) are known, one can estimate the length of time (t_r) required for the wave to pass through a point as

$$t_r = \frac{l}{L_h}.$$

As discussed earlier, sediment waves that constitute disturbances of the type discussed here range in length from several hundred meters to several kilometers in length (Meade 1985, Madej and Ozaki 1996), and annual travel distances range from hundreds to thousands of meters per year. Therefore, in large channels with relatively small inputs t_r should be less than one year, and in small channels with relatively large inputs t_r could be ten years or more. For sediment waves of equivalent length, recovery time is inversely related to annual travel distance.

It should also be noted that this recovery time only applies to reaches where bed load waves are presently in transit. Recovery time of an entire channel network from a single large sediment

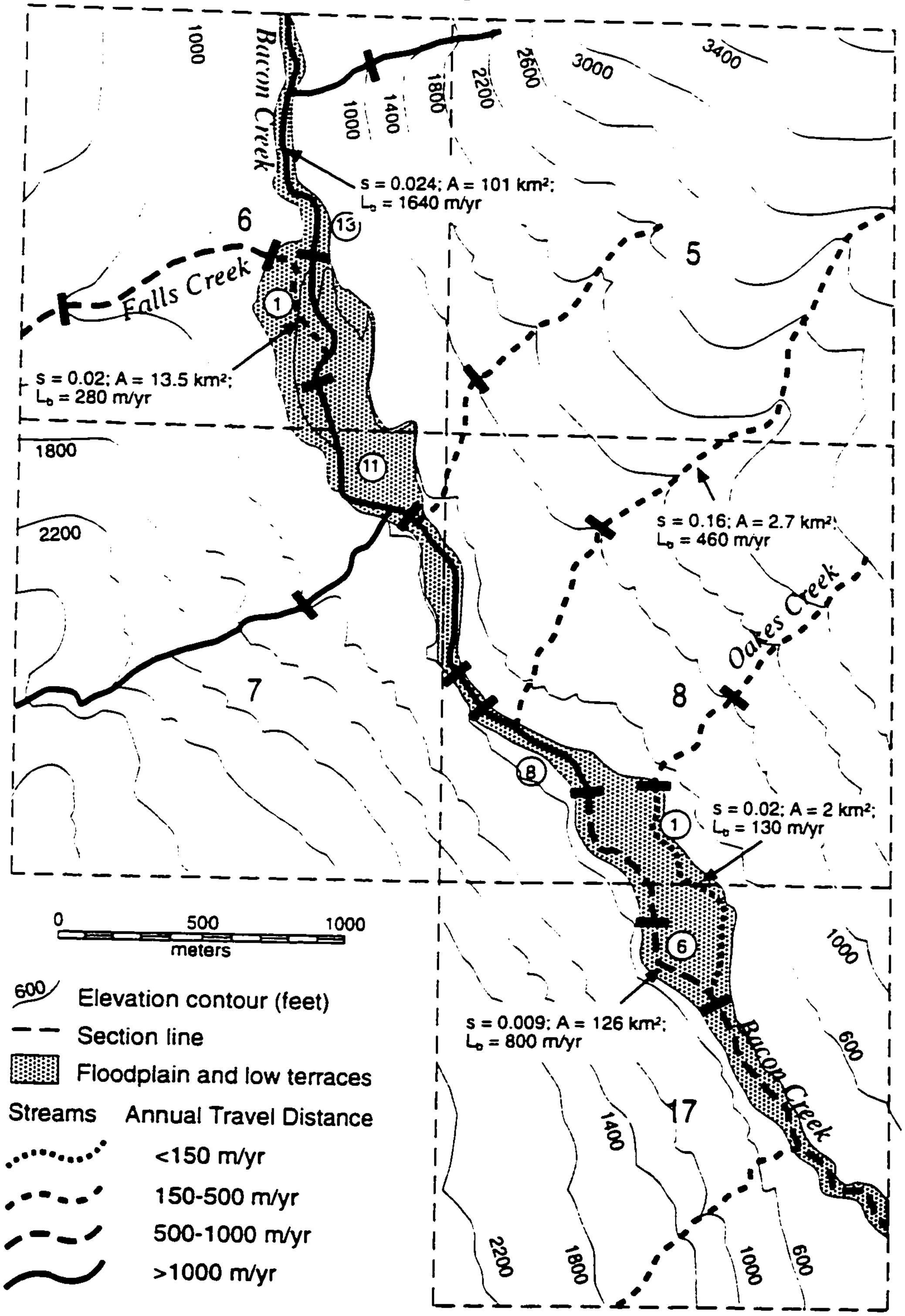


Figure 4-9. Annual travel distances of bed load sediment (based on $L_b = 94 + 85\Omega$) and relative recovery potentials for streams in a five square mile area of Bacon Creek sub-basin. Small terrace tributary reaches such as Reach 1 of Oakes Creek have the slowest annual travel distances and lowest recovery potential. Relatively steep and large mainstem reaches such as reaches 8-13 of Bacon Creek have the greatest annual travel distances and greatest recovery potential.

input may take several decades when downstream reaches are included in the analysis. In this situation, recovery from disturbance at a single point should remain between one and ten years, but the disturbance moves gradually downstream. Therefore, upstream reaches may have "recovered" from a sediment wave while reaches far downstream have not yet been disturbed. Other authors have shown that sediment may take several decades to pass through a channel network in such cases (Kelsey 1982b, Madej 1992).

Influence of riparian vegetation and LWD on recovery time

Other factors may affect recovery time by altering either the rate of sediment transport or the rate at which sediment is transferred from the channel to floodplain storage. The primary factors that may alter recovery time are the condition of riparian vegetation and the abundance of in-channel LWD. LWD in stream channels can increase recovery time by slowing the transport of sediment through the reach. Riparian vegetation may inhibit channel widening, which decreases the amount of sediment recruited to the channel from floodplain storage and decreases the area of denuded floodplain. Reduced channel widening may decrease recovery time by decreasing the quantity of sediment that must be transferred out of the reach, and by eliminating the time required to re-establish floodplain vegetation. Larger riparian vegetation will also inhibit lateral migration of channels, which also will help decrease the amount of sediment recruited to the channel from the floodplain.

Mature riparian forests may also reduce recovery time by increasing LWD recruitment as a channel widens or migrates. Such LWD recruitment is a critical component of LWD jam formation, which can protect bar surfaces from high energy flood flows and enhance recovery of riparian vegetation (Collins et al. 1994, Abbe and Montgomery 1996). This recovery mechanism increases the rate at which sediment is transferred from in-channel storage to floodplain storage. Under natural conditions, increased sediment supply to reaches bordered by mature forests initiates a cycle that promotes recovery of riparian vegetation. Increased sediment supply first causes bank erosion, which increases LWD recruitment and debris jam formation, and which in turn promotes the revegetation of bar and floodplain surfaces.

After logging of riparian forests, this cycle is interrupted. Under logged conditions, channel widening does not increase LWD recruitment because the trees have been removed, and part of the natural recovery process is eliminated (Collins et al. 1994). Without stable debris pieces or jams the channel is more prone to lateral migration, which continually reactivates channel-stored sediment and inhibits the recovery of riparian vegetation. At a minimum, recruitment of woody debris will not be restored until the quantity of channel-stored sediment has been reduced and young forests are re-established on the floodplain.

Landscape-level patterns of channel response to changing sediment supply

By modeling sediment supply and transport in a large channel network over several thousand years, Benda (1994) showed that steep reaches with high transport capacity were very resilient, with only short periods of aggradation interspersed with long periods of low sediment storage. These reaches have high sediment transport capacity and would be considered to have a high recovery potential. Channels of intermediate slope and size tended to have longer periods of aggradation due to reduced sediment transport capacity (i.e., reduced recovery potential). Larger channels had higher transport capacity (i.e., higher recovery potential), and therefore showed little change in sediment storage at any time.

Stream channels in different parts of a large channel network respond dissimilarly to changes in sediment supply. The general pattern of responses to large sediment inputs in a channel network was described by Benda (1994), based on the physical characteristics of an Oregon Coast Range river basin. Benda's modeling of sediment supply and routing over several thousand years showed that steeper alluvial reaches near the source of sediment tend to have high stream power, and that aggradational events are relatively few in number and of short duration. Farther downstream, moderate-slope to low-slope reaches tend to have low stream power, yet they can directly receive large sediment inputs via mass wasting. Hence, aggradational events can be severe and long lasting in these mid-order channels. However, in larger low-gradient reaches where most sediment is received by fluvial transport from upstream reaches, higher stream power, attenuation of bed load waves, and particle attrition tend to reduce the magnitude of aggradational events.

Such differences tend to be systematically related to drainage area in Benda's Oregon Coast Range study area, but they may be less systematic in a younger landscape such as the North Cascades of Washington State. In our study area, a more complex bedrock geology and younger sediments deposited during Pleistocene continental glaciation have strong influences on the location of sediment source reaches relative to higher order streams, resulting in some reaches of larger channels having the potential to directly receive large sediment inputs. Thus, altered sediment supply in the study area can have direct effects on both large and small channels in drainage networks of the study area.

As Benda (1987) showed, sediment transport in the steepest 1st and 2nd order channels may be dominated by debris flow (Table 4-3). These channels typically have slopes greater than 18% (Benda 1985), and channel beds are typically dominated by colluvium or bedrock (Benda 1994, Montgomery and Buffington 1997). Sediment delivered to these channels by soil creep or small mass wasting events is typically stored for long periods of time, and then released periodically by debris flow (Benda and Dunne 1987). These channels are therefore considered transport-limited (sediment supply greater than sediment transport capacity).

Farther down the channel network, channel beds are dominated by bedrock or very coarse alluvium, indicating an abrupt change to supply-limited channels (sediment supply lower than transport capacity). These step-pool and cascade channels have slopes between about 3% and 20% (Montgomery and Buffington 1997), and bed morphologies may be boulder cascades or a series of steps formed by ordering of boulders into cross channel accumulations (e.g., Whitaker 1987, Chin 1989). Sediment in these channels is typically transported by fluvial means, with finer materials being frequently transported over larger stable material (Montgomery and Buffington 1997). Larger particles that form the bed are typically moved only in the largest floods. Large inputs of coarse sediment to such channels is transported in waves (Whitaker 1987), and the bed aggradation associated with large inputs is short-lived (Benda 1994). Therefore, cascade or step-pool bed morphology is most commonly observed, reflecting a predominantly supply-limited sediment transport condition. Low-gradient channels tend to transport sediment more slowly, and therefore exhibit more pronounced morphological responses to increased sediment supply (Benda 1994, Montgomery and Buffington 1997). These reaches have slopes less than 4%, and may exhibit pool-riffle or plane bed morphologies (Montgomery and Buffington 1997). Aggradational events in these reach types may last decades in smaller channels, but may be relatively rare in the largest channels due to particle attrition and attenuation of bed load waves (Benda 1994).

Table 4-3. Typical magnitude and duration of bed load aggradation in different channel types in western Washington. Channel slope and morphology descriptions based on Montgomery and Buffington (1997). Typical annual travel distance (L_b) based on the empirical relationship L_b = 94 + 85ω, described in text. Typical depth and duration of aggradational events based on Benda (1994).

Stream order	**Typical channel slope and morphology**	**Typical annual travel distance**	**Typical depth of bed-load aggradation**	**Typical duration of aggradational events**
1-2	>18%; colluvium, bedrock	NA	NA	long-term storage of colluvium; periodic debris flow erosion of sediments
3-4	8-20%; boulder cascade	600 m/yr	<1 m	<20 years
	3-8%; step pool	250 m/yr	<1 m	10-100 years
	<4%; pool-riffle, plane-bed, forced pool-riffle	200 m/yr	<1 m	10-100 years
5-6	8-20%; boulder cascade	1200 m/yr	1-2 m	<10 years
	3-8%; step pool	500 m/yr	1-2 m	<10 years
	<3%; pool-riffle, plane-bed, forced pool-riffle	850 m/yr	1-2 m	< 10 years
>7	<1%; pool-riffle, forced pool-riffle	1700 m/yr	<1 m	<10 years

Chapter 5. Disturbance and recovery of LWD recruitment processes

Changes in riparian forest conditions produced by fire, stream bank erosion, or forest management can alter large woody debris (LWD) recruitment to streams, which in turn alters the habitat characteristics of streams (Grette 1985, Andrus et al. 1988, Carlson et al. 1990, Bilby and Ward 1991, Ralph et al. 1994). Altered habitat characteristics of streams typically cause changes in the salmonid carrying capacity of a stream (e.g., McMahon and Hartman 1989, Hicks et al. 1991). As with changes in sediment supply, understanding the linkages between changes in riparian forests and habitat characteristics allows managers to identify causes of degraded habitat conditions and to devise restoration actions having greater likelihood of success (Frissell and Nawa 1992, Beechie et al. 1996, Roper et al. 1997).

Clearcut logging reduces the potential of a riparian forest to recruit LWD for several decades (Grette 1985, Murphy and Koski 1989, Bilby and Ward 1991). Because depletion of instream LWD continues during the period of low recruitment, LWD abundance declines initially (Grette 1985) and remains low between 50 and 100 years after logging (Murphy and Koski 1989). Recovery of LWD recruitment depends in part on the species composition of new stand. For example, alder LWD can be recruited as early as 25 years after logging, whereas recruitment of small conifer LWD typically begins after about 50 years (Grette 1985, Andrus et al 1988). Recruitment of large conifer LWD (>60 cm) does not begin until about 75 years after logging (Murphy and Koski 1989).

A change in LWD abundance alters fish habitat characteristics such as pool spacing, pool area, and pool depth (Bilby and Ward 1989, Montgomery et al. 1995, Abbe and Montgomery 1996). In general, decreased LWD abundance causes decreased pool abundance and depth (Montgomery et al. 1995, Beechie and Sibley 1997). However, a given change in LWD abundance has a greater effect on pool abundance in moderate-slope channels (slope between 0.02 and 0.05) than in low-slope channels (slope< 0.02) (Beechie and Sibley 1997). The effect of LWD abundance on spawning habitats is less clear. Several researchers have shown that individual LWD pieces store sediment in steeper stream channels (Megahan 1982, Bilby and Ward 1989, Bilby and Ward 1991, Nakamura and Swanson 1992). However, reach-level relationships between LWD abundance and spawning habitat abundance have not been found, despite the fact that individual debris accumulations trap sediment (House and Boehne 1985, Bilby and Ward 1989, Beechie and Sibley 1997).

The primary objectives of this chapter are to describe how the supply of LWD from the existing riparian zone differs from that of riparian zones which likely existed prior to extensive logging (degree of disturbance) and the rate at which LWD recruitment and in-channel LWD can be restored to "natural" conditions (recovery time). The first part of this chapter addresses land-use effects on LWD recruitment at the site and landscape levels. It first describes patterns of disturbance and recovery of LWD recruitment at the site levels. Disturbance to LWD recruitment is primarily a function of alterations in the size, density, or species composition of trees in the riparian forest. Recovery of LWD recruitment is mainly a function of the species recolonizing a site and the size of LWD that is sufficient to significantly alter channel morphology (e.g., to form pools). At the landscape level, disturbance and recovery are primarily functions of changes in the distribution of site-level stand ages and LWD recruitment potentials. These distributions are a function of the disturbance regime imposed on the landscape, which are illustrated in this dissertation by two wildfire regimes that represent natural conditions and three forest management scenarios that represent current conditions.

The second part of this chapter describes the magnitude and duration of effects of changes in LWD abundance on pool characteristics at the site and landscape levels. It first describes the cause-effect relationships between LWD abundance and pool characteristics, which provide the last linkage between land-use effects on LWD recruitment and habitat characteristics in the stream. Secondly, it describes recovery rates of LWD abundance and pool characteristics by two pathways: the red alder pathway and the Douglas fir pathway. Because recovery rates are a function of the species recolonizing a site, this section retraces some aspects of the duration of land-use effects on LWD recruitment described in this first part of this chapter. This section first describes two models that are used to predict the changes in riparian condition, LWD abundance, and pool-riffle morphology over time. It then describes LWD recruitment recovery pathways for red alder and Douglas fir at the site level, as well as differences in recovery rates by channel width and slope. Finally, this section describes landscape-level patterns of changes in pool abundance that have resulted from riparian disturbance.

Magnitude and duration of land-use effects on LWD recruitment

For the purposes of this chapter, logging of riparian forests is considered the main cause of disturbance to LWD recruitment. On a per area basis, road building is a relatively minor component of disturbance to riparian forests in the study area. Channel widening, migration, and avulsion resulting from increased sediment supply or flooding are also disturbance mechanisms, but their causes are external to a stream reach and its associated riparian forest (Chapter 4). In such cases, sediment supply to the reach must recover before LWD recruitment can begin to recover via the pathways described in this chapter. The recovery pathway of LWD recruitment is then considered to be generally similar to that of a post-logging scenario.

This section first describes the magnitude and duration of logging related disturbances to LWD recruitment at the site scale. It then describes how site-level effects may be expressed at the landscape scale under differing disturbance regimes. Two wildfire regimes represent natural conditions, and three forest management scenarios represent present-day conditions. Comparisons among disturbance regimes are expressed in terms of percentage of riparian stands less than a threshold age under each regime, and as the relative amount of LWD recruitment under each regime. For each case, the threshold age is based on the stand age at which recruitable hardwood or conifer LWD is larger than a specified minimum diameter.

Site-level disturbance and recovery of LWD recruitment processes

After clearcut logging of a riparian forest, there is no standing LWD available for recruitment and LWD recruitment is zero. Logging to the banks of streams was common in the study area until approximately the mid- 1980s, but now typically occurs only along smaller and steeper streams that do not support fish populations. Where stream buffers are left during logging operations, authors generally predict that some LWD recruitment will be maintained after logging (e.g., Murphy and Koski 1989, McDade et al. 1990) or will increase after removal of the surrounding forest (e.g., Beechie 1994, Mobbs and Jones 1995). However, even along fish bearing streams regulation buffers of 25 to 50 feet (7.5 to 15 m) (e.g., Washington Forest Practices Rules) are insufficient to provide LWD equivalent to an uncut stand (Van Sickle and Gregory 1990). Recruitment of LWD from the post-logging stand does not begin until several decades after the disturbance (Grette 1985, Andrus et al. 1988, Murphy and Koski 1989).

Post-fire LWD recruitment to a stream channel may not decrease from pre-fire levels, and may even increase after the disturbance (Reeves et al. 1995). Along smaller streams, fires appear to have been the dominant disturbance mechanism (Agee 1988), and stands were typically conifer dominated (Gannet 1899). Fires appear to have been less frequent along larger rivers (i.e., in valley bottoms) due to higher humidity and fuel moisture, as well as to a greater proportion of less flammable deciduous vegetation (Gannett 1899, Agee 1988).

LWD recruitment after fire was probably highly variable among sites because several factors contribute to LWD recruitment during the post-fire period. A stand-replacing fire would leave standing dead trees next to the stream, tending to increase the rate of recruitment after fire because of the increased mortality of the riparian forest. Standing dead trees may also have been more vulnerable to windthrow because of decaying root strength and greater exposure to wind by removal of the surrounding forest, yet reduced areas of needles and branches exposed to winds would tend to decrease vulnerability to windthrow. Decaying root strength may also have increased the probability of bank erosion, which would tend to increase LWD recruitment after a fire. Although it is unclear which factors may have predominated, most factors lean toward increasing LWD recruitment within the first few years to decades after a fire. LWD recruitment presumably decreased dramatically within a few decades as most dead trees had fallen and the new riparian stand was becoming established.

For this study, LWD recruitment rates after a riparian forest is either burned or logged are presumed to be zero until recruitment from the new forest begins. That is, the time to first LWD recruitment from a post-fire stand is presumed similar to that after logging as described by Grette (1985), Andrus et al. (1988), and Murphy and Koski (1989). These assumptions may result in underestimating LWD recruitment after a fire, but should be appropriate for estimating post-logging recruitment. Therefore, the assumptions may cause an underestimate of the difference in LWD recruitment between a fire regime and a timber harvest regime. They will not affect estimates of the present-day recovery rates among sites in the study area because there are no fire-disturbed riparian forests at present.

Grette (1985) and Andrus et al. (1988) showed that the abundance of post-logging alder LWD in channels begins to increase after about 25 years, with peak abundance of alder from the post-logging stand at about 70 years after disturbance (Figure 5-1a). These same studies also showed abundance of conifer LWD > 10 cm diameter starting to increase at about 60 years after disturbance, and continuing to increase until at least 130 years after disturbance. However, Murphy and Koski (1989) showed that abundance of larger conifer LWD (> 60 cm diameter) does not begin to increase substantially until after about 90 years after disturbance, and continues to increase until at least 250 years after disturbance.

Because different species or sizes of LWD can have different recovery rates, describing recovery of LWD recruitment at a site requires identification of important LWD species or sizes. Where such LWD characteristics have been identified, the point in time where abundance of suitable LWD begins to increase can be considered a threshold of recovery for LWD recruitment. For example, the recovery threshold for recruitment of LWD > 10 cm diameter is 25 years if species is not an important factor (T_1 in Figure 5-1a). However, the recovery threshold for recruitment of conifer > 10 cm diameter is 60 years (T_2), and the recovery threshold for recruitment of conifer > 60 cm diameter is about 90 years (T_3). For this dissertation, choice of recovery thresholds will be determined primarily by the size of LWD required to influence pool formation (e.g., Beechie and Sibley 1997) or to anchor LWD jams (e.g., T. Abbe, Department of Geological Sciences, University of Washington, Seattle, unpublished data).

Recovery of the *process* of LWD recruitment is not equivalent to recovery of the *abundance* of LWD. Recovery of LWD abundance in a stream channel is gradual (Figure 5-1a), and may begin before the pool-forming size threshold is reached (with recruitment of small woody debris). Once the threshold LWD size is attained, LWD abundance will continue to increase gradually for many decades, and the size of recruited LWD will change gradually throughout the recovery period. Recovery of the process as defined previously is a threshold relationship in which the process is not active until the threshold is reached, and then becomes fully active within a short period of time once the threshold is attained. Therefore, a disturbance-recovery curve for the relative LWD recruitment rate (the "disturbed" LWD recruitment rate divided by the "recovered" LWD recruitment rate) will drop to zero after logging and remain zero until the recovery threshold is reached, after which the rate rapidly rises to 1 (Figure 5-1b). Note that this recovery model ignores changes in the size of recruited LWD over time once the pool-forming size threshold is attained.

Landscape-level effects of altered disturbance regime on LWD recruitment

As with the landscape-level analysis of sediment supply, the proportion of the landscape below a specified threshold may be used to estimate the landscape-level effect of an altered disturbance regime on LWD recruitment. For example, one may estimate the proportion of riparian forests that have trees too small to effectively create pools in a stream. Such a threshold is spatially variable because different sized streams require different sizes of LWD

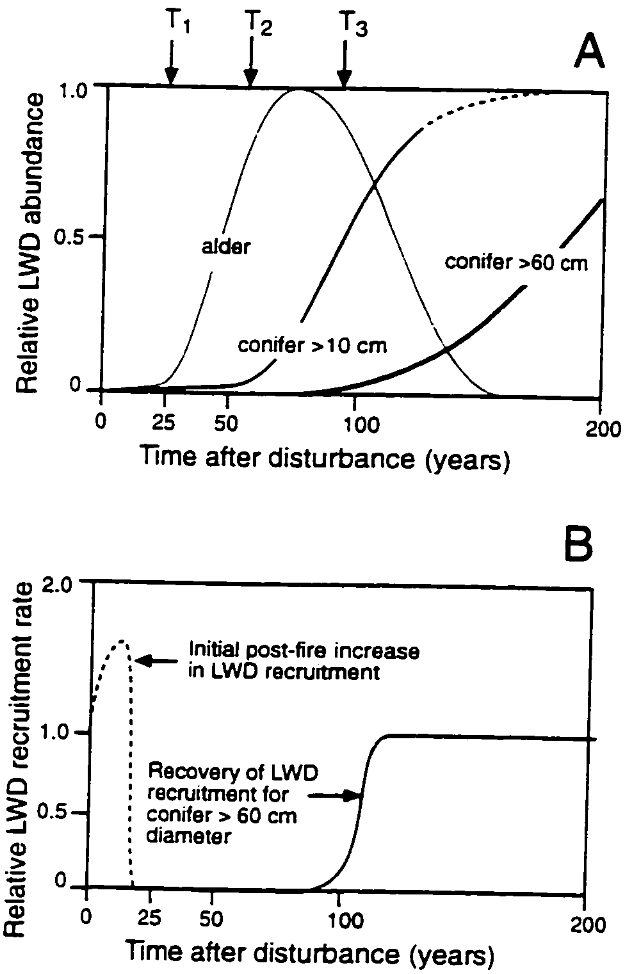


Figure 5-1. (A) Examples of recovery thresholds for LWD recruitment based on measurements of in-channel LWD abundance. Curves are composites based on Grette (1985), Andrus et al. (1988), and Murphy and Koski (1989). T_1 is the threshold for recovery of alder LWD >10 cm diameter; T_2 is the threshold for recovery of conifer LWD >10 cm diameter; T_3 is the threshold for recovery of conifer LWD >60 cm diameter. The selection of a recovery threshold for a particular site depends primarily on the size of LWD required to influence pool formation in the channel. (B) Hypothetical recovery of LWD recruitment to a large stream channel after a disturbance. Solid line represents recovery after clearcutting for LWD > 60 cm diameter. Dashed line illustrates a possible increase in LWD recruitment for several years after a fire due to increased recruitment of standing dead trees.

to effectively form pools. Therefore, the effect of an altered disturbance regime on LWD recruitment will vary by watershed because each channel network has varying proportions of its stream length in channels of different sizes. The purpose of this section is not to analyze specific watersheds, but to illustrate the range of effects that forest management disturbance regimes have had on LWD recruitment compared to natural disturbance regimes.

To illustrate this variability across a landscape, this section describes the effects of two natural scenarios (fire regimes for western hemlock and silver fir) and three forest management scenarios (harvest rotations of 40, 60, and 80 years) for each of three different LWD recovery thresholds. Under natural conditions, the characteristics of riparian forests vary considerably due to disturbances such as fire, avalanches, and channel migration (Agee 1993, Oliver et al. 1985). For the purpose of this illustration, the examples focus only on disturbance by fire, which is typical of smaller channels in a drainage network (Agee 1988). The assumption may not always be appropriate for larger channels, but data from nearby Deer Creek (tributary to North Fork Stillaguamish) indicates that stream channels at least as large as 45 meters wide are rarely disturbed by channel migration or avulsion in the absence of a disturbance to riparian vegetation. Most channels considered in this study are less than 45 meters wide.

Each LWD recovery threshold represents a different point in the succession and growth of a riparian forest, along with the associated LWD species and sizes at each time (Figure 5-1a). Threshold T_1 is the point at which recruitment of alder LWD > 10 cm diameter begins (25 years after disturbance); threshold T_2 is the point at which recruitment of conifer LWD > 10 cm diameter begins (60 years); and threshold T_3 is the point at which recruitment of conifer LWD > 60 cm diameter begins (90 years).

Along smaller streams where small LWD (including alder) can form pools, the long term average percentage of forests less than 25 years old is 12% under the western hemlock fire regime, and 7% under the silver fir fire regime (Table 5-1). However, under a forest management regime with an average rotation length of 60 years, more than 40% of the area would be less than 25 years old, and variation in the rotation length could increase the percentage to as much as 63% of the area. Landscape-averaged recruitment rates of pool-forming LWD under a rotation length of 60 years would then be an estimated 66% of the rate under the western hemlock fire regime (Table 5-2). Variation in harvest rotations down to 40 years could reduce that rate to as little as 42% of the rate under the western hemlock fire regime. Along larger streams where LWD > 60

Table 5-1. Percentage of riparian stands below threshold age (T) under typical natural and management disturbance regimes. See text for explanation of fire and management regimes. Estimates assume no regulatory protection of riparian stands, as was the case until the mid-1980s. T_1 is the age at which recruitment of hardwood LWD >10 cm diameter begins; T_2 is the age at which recruitment of conifer LWD >10 cm diameter begins; T_3 is the age at which recruitment of conifer LWD >60cm diameter begins.

Disturbance regime	Percentage of area below threshold age (T)		
	T_1 = 25 years	T_2 = 60 years	T_3 = 90 years
Silver fir fire regime	7	15	22
Western hemlock fire regime	12	26	36
80-yr harvest rotation	31	75	100
60-yr harvest rotation	42	100	100
40-yr harvest rotation	63	100	100

Table 5-2. Landscape-level relative rates of LWD recruitment under typical natural and management disturbance regimes. The base rate is the average LWD recruitment rate for the western hemlock zone with the long-term average percentage of stands younger than threshold age. Relative recruitment rates are calculated with the assumption that site-level LWD recruitment rates are similar in all five disturbance regimes.

Disturbance regime	Relative LWD recruitment rate		
	T_1 = 25 years	T_2 = 60 years	T_3 = 90 years
Silver fir fire regime	1.06	1.15	1.22
Western hemlock fire regime	1.00	1.00	1.00
80-yr harvest rotation	0.78	0.34	0
60-yr harvest rotation	0.66	0	0
40-yr harvest rotation	0.42	0	0

cm diameter is required to form pools and threshold age is 90 years, a much higher proportion of the landscape would be below threshold age under each disturbance regime (Table 5-1). For a threshold age of 90 years, the silver fir and western hemlock fire regimes would have 22% and 36% of stands below threshold, respectively, and all three forest management regimes would have 100% of stands younger than 90 years. Under these scenarios, none of the forest management regimes would produce LWD of pool-forming size (Table 5-2).

Recruitment of alder LWD > 10 cm diameter could begin in as little as 25 years. However, small streams in western Washington typically require LWD of at least 20 cm diameter to create pools (Beechie and Sibley 1997), and the average size of LWD in small streams in old-growth typically exceeds 35 cm (Bilby and Ward 1989). Therefore, recruitment rates of functional LWD are not expected to recover until 40 or 50 years after disturbance even for small streams. For larger streams (channel width > 20 m) recovery of LWD recruitment is expected to be longer than the 90-year example shown here because smallest pool-forming LWD in larger streams exceeds 60 cm diameter (Beechie and Sibley 1997), and the average diameter typically exceeds 70 cm (Bilby and Ward 1989).

Magnitude and duration of effects of altered LWD recruitment on stream channels

This section is separated into three parts: (1) the effect of altered LWD abundance on channel morphology (primarily pool formation) at the site level, (2) the rates at which LWD abundance and pool formation recover from altered riparian conditions at the site level, and (3) landscape-level patterns of channel response to changes in LWD recruitment. The first part explains the mechanisms responsible for changes in channel morphology, focusing on relationships between LWD abundance and pool-formation and the influence of channel width and slope on these relationships. The second part describes the recovery of LWD abundance and pool characteristics using two computer models: an individual tree growth model that simulates riparian forest development and an LWD recruitment and function model that simulates treefall, LWD input, LWD depletion, and pool formation. Because recovery of LWD abundance and pool characteristics is dependent on the recovery pathway (i.e., the species recolonizing a site), this part revisits the recovery of riparian forests and LWD recruitment described in the previous section. The third part of this section describes landscape-level patterns of channel response to LWD recruitment based on channel size and slope.

Magnitude of effects of altered LWD recruitment on channel morphology – reach scale

Increases in LWD abundance generally result in increases in number of pools (usually quantified as pool spacing) or pool area (Bilby and Ward 1989, Montgomery et al. 1995, Beechie and Sibley 1997). In channels with slope less than about 0.04, pool spacing decreases as LWD abundance increases, but the relationship between LWD and pools varies with channel slope (Montgomery et al. 1995, Beechie and Sibley 1997). Pool spacing is less sensitive to LWD abundance in low-slope channels (<0.02) than in moderate-slope channels (0.02 to 0.05) (Beechie and Sibley 1997). In low-slope channels without LWD, pools are formed by lateral scour at banks at a typical spacing of 5 to 7 channel widths per pool (Leopold et al. 1964). By contrast, pools do not appear to form in moderate-slope channels without LWD because LWD or other obstructions are required to force flow convergence and initiate scour of a pool (Beechie and Sibley 1997). Pool spacing in steeper channels appears to be independent of LWD abundance, at least partly due to the fact that boulder step-pools tend to form at a spacing of 1-4 channel widths (Montgomery et al. 1995).

Pool area also generally increases with increasing LWD abundance (Bilby and Ward 1989), although pool area also is less sensitive to LWD abundance in low-slope channels than in moderate-slope channels. Furthermore, it is possible that increased LWD abundance in low-slope channels does not contribute to increased pool area, as suggested by one study that shows channel width is a better predictor of pool area in low-slope channels than is LWD abundance (Beechie and Sibley 1997). Again, free-formed pools tend to maintain a pool-riffle morphology in low-slope channels even when LWD is not present (Montgomery et al. 1995).

For this study, these relationships must be quantified to allow prediction of pool characteristics as LWD abundance changes. The form of the predictive equations was governed primarily by two constraints: (1) the types of data available, and (2) the types of LWD outputs that the LWD recruitment model can produce with greatest confidence. In this case, the greater constraint was low confidence in the recruitment model's ability to estimate LWD volumes. The main difficulty is that piece breakage has not been quantified, and estimating the volume of individual LWD pieces is largely a function of piece length. Therefore, all of the equations for predicting pool characteristics from LWD abundance are based on number of LWD. Data for describing these relationships are taken primarily from Montgomery et al. (1995), Beechie and Sibley (1997), and George Pess (stream ecologist, Tulalip Tribes, unpublished data). Each data set has channel slope, channel width, and number of LWD. All three data sets included number of pools, and all were used in developing the LWD and pool-spacing equation. Pool areas were not included in Montgomery et al. (1995), so pool area equations use only the data of Beechie and Sibley (1997) and G. Pess (unpublished data).

Prediction equations for pool spacing

For pool spacing, I stratified channels into three slope classes based on the relative sensitivity of pool formation to LWD abundance. In low-slope channels (here defined as slope $\leq$ 0.01), pool spacing is a function of LWD/m, but at low LWD abundance pool spacing rarely exceeds 5 to 7 channel widths per pool because free-formed pools maintain the pool-riffle morphology (Montgomery and Buffington 1997). Moderate-slope channels (0.01 < slope $\leq$ 0.04) are more sensitive to LWD abundance because LWD is required to initiate pool formation. Pool spacing at low LWD abundance can often exceed 10 channel widths between pools (w_{bkf} /pool) in these channels. In steeper channels (slope > 0.04) cross-channel accumulations of cobbles and boulders tend to dominate pool formation, with spacing between 1 and 4 channel widths per pool (e.g., Chin 1989). LWD does not appear to alter pool spacing in these channels (Montgomery et al. 1995).

At slope < 0.01, pool spacing declines exponentially as LWD/m increases, reaching a low of about 0.6 w_{bkf} /pool (Figure 5-2, Table 5-3). When LWD/m is less than about 0.03, pool spacing is set at the fixed value of 6. This fixed value is based on the empirical 5 to 7 w_{bkf} /pool (Leopold et al. 1964), and represents the spacing of free-formed pools in low-slope channels with low LWD abundance. At slope between 0.01 and 0.04, pool spacing also declines exponentially with increasing LWD abundance, but pool spacing is typically longer than in low-slope channels with a similar LWD abundance because there are virtually no free-formed pools in the absence of LWD. Hence, pool spacing can exceed 10 w_{bkf} /pool when LWD abundance drops below about 0.05 LWD/m. At very low LWD abundance, pool spacing is fixed at 15 w_{bkf}/pool, which approximates the longest pool spacing measured by Montgomery et al. (1995). For channels with slope greater than 0.04, LWD abundance does not affect pool spacing and pool spacing is fixed at the average spacing of 1.2 w_{bkf}/pool.

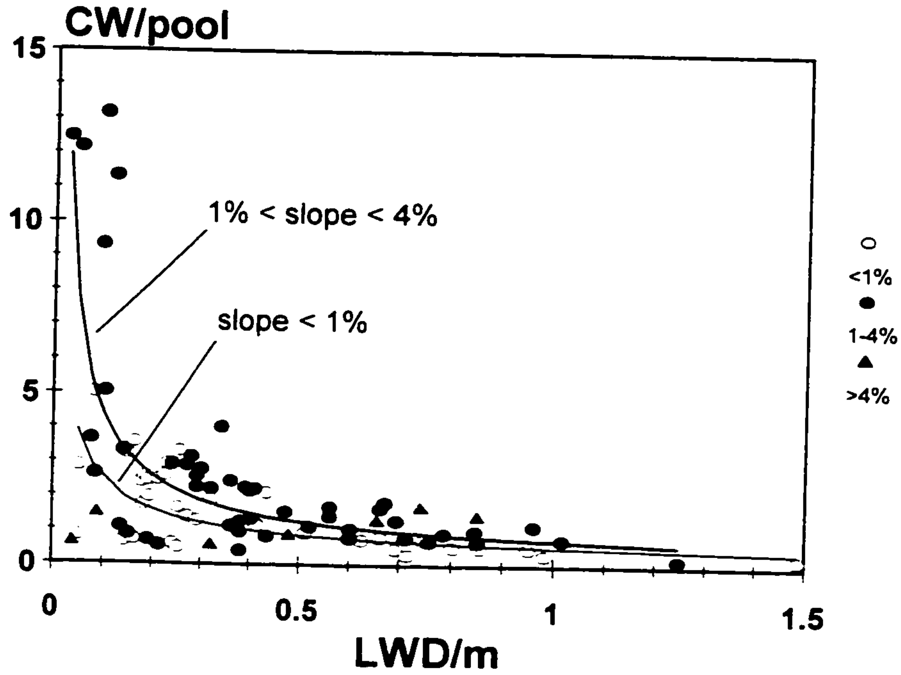


Figure 5-2. Pool spacing as a function of LWD per meter and channel slope class. Data are from Montgomery et al. (1995), Beechie and Sibley (1997), and G. Pess (unpublished data). For slope < 0.01, regression equation is w_{bkf}/pool = $0.56(\text{LWD/m})^{-0.65}$. For slopes between 0.01 and 0.04, regression equation is w_{bkf}/pool = $0.74(\text{LWD/m})^{-0.78}$. For channels with slope > 0.04 there is no relationship with LWD/m; average w_{bkf}/pool = 1.2 (stdev = 0.4). See Table 5-3 for other regression statistics.

Table 5-3. Prediction equations for pool spacing and percent pool based on LWD/m, LWD/m^2, or bankfull channel width (w_{bkf}). Regressions for pool spacing include data from Montgomery et al. (1995), Beechie and Sibley (1997), and G. Pess (unpublished data). Regressions for percent pool include data from Beechie and Sibley (1997), and G. Pess (unpublished data).

Output	**Slope**	**Equation**	r^2	**n**	**p**
Pool spacing	$s \leq 1\%$	$PS = 0.56(LWD/m)^{-0.65}$	0.45	36	< 0.001
	$1\% < s \leq 4\%$	$PS = 0.74(LWD/m)^{-0.78}$	0.54	48	< 0.001
	$s > 4\%$	PS = 1.2 (st.dev.= 0.4)		7	
Percent pool	$s \leq 1\%$	$PP = -2.2(w_{bkf}) + 79$	0.49	17	0.002
	$1\% < s \leq 4\%$	$PP = -15(slope) + 328(LWD/m^2) + 57$	0.53	22	0.0005
	$s > 4\%$	PP = 27 (st.dev.= 5)		4	

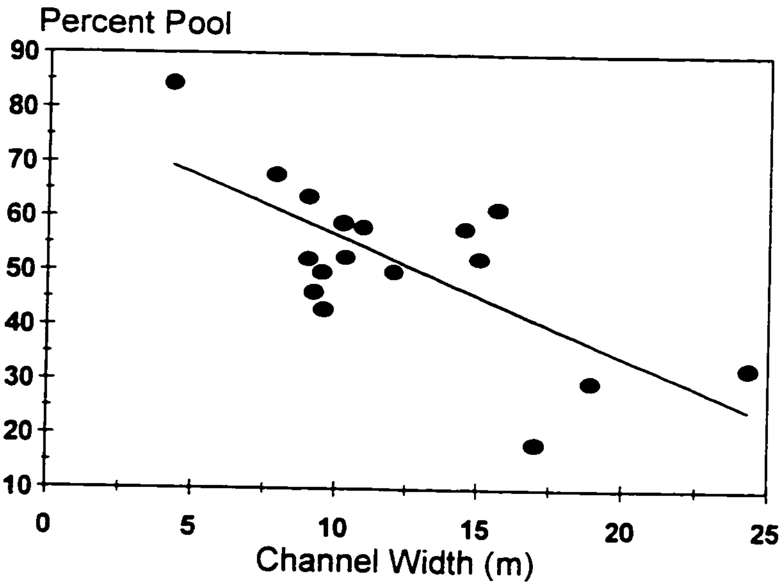
Prediction equations for percent pool

Pool area (expressed as percent pool) has been most commonly related to LWD volume per unit channel area (e.g., Bilby and Ward 1989). However, Beechie and Sibley (1997) found that in low-slope channels, channel width was a better predictor of percent pool than was LWD volume. This suggests that fewer free-formed pools in low-slope channels may account for as much total pool area as a greater number of LWD-formed pools. That is, at slopes < 0.01 increasing the number of pools formed by LWD does not increase total pool area, but creates more pools with smaller individual areas. For low-slope channels, the combined data of Beechie and Sibley (1997) and Pess (unpublished data) showed that a negative linear relationship adequately represents the relationship between percent pool and channel width, at least out to a channel width of about 25 m (Figure 5-3a, Table 5-3).

Beechie and Sibley (1997) found that percent pool in moderate-slope channels was strongly related to LWD volume/m^2 channel area, and that slope and LWD volume/m^2 were both significant predictors of percent pool in a multiple linear regression. However, the LWD recruitment model used here does not accurately estimate LWD volume/m^2. Therefore, I substituted number of LWD per unit channel area (LWD/m^2) for LWD volume/m^2, and generated a new regression relationship based on data from Beechie and Sibley (1997) and G. Pess (unpublished data). Compared to the original regression relationship, the revised multiple regression relating percent pool to slope and LWD/m^2 has the same general pattern of percent pool decreasing with increasing channel slope and increasing with increasing LWD/m^2 (Figure 5-3b). It also had a similar correlation coefficient (Table 5-3).

Use of prediction equations

The first use of the prediction equations for pool spacing is to describe how changes in LWD/m affect number of pools in a reach as described in the preceding paragraphs. A second use of these equations will be to help describe changes in pool spacing over time as a function of LWD input and depletion (see next section and Appendix B). Similarly, the prediction equations for percent pool are first used to describe how altered LWD abundance affects percent pool (preceding paragraphs), and secondly to describe changes in percent pool over time (next section and Appendix B). In order to provide some context for the predicted pool spacing and percent pool, the following two paragraphs describe the approximate ranges of pool spacing and percent



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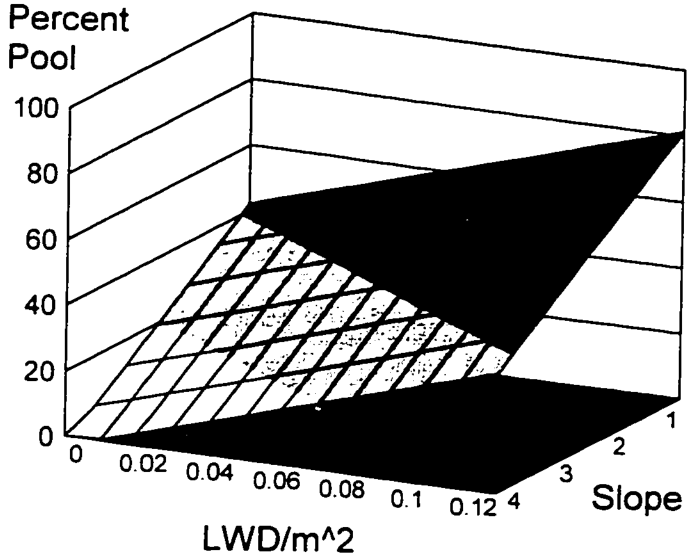


B

Figure 5-3. (A) Relationship between percent pool and channel width for low-slope channels (<0.01). The regression equation is (Percent Pool) = $-2.2(w_{bkf}) + 79$. (B) Percent pool as a function of slope and LWD/m^2. The regression equation is (Percent Pool) = $-15(\text{slope}) + 328(\text{LWD}/m^2) + 57$. Regression statistics for both equations are in Table 5-3. Both figures are based on data from Beechie and Sibley (1997) and G. Pess (unpublished data).

pool in streams in old-growth forests.

Where LWD/m decreases due to a disturbance, pool spacing will increase exponentially as a function of the decrease in number of LWD. LWD and pool spacing are considered outside the range of natural variability when LWD/m is less than 0.4 (where LWD is classified as debris > 10 cm diameter and > 1 m long) and pool spacing is greater than 2 (where pools are classified as any distinct morphological depression) (Montgomery et al. 1995). These values are characteristic of Pacific Northwest streams that have been affected by logging, agriculture, or urbanization. LWD and pool abundance are considered within the range of natural variability when LWD abundance is greater than about 0.4 LWD/m and pool spacing is less than about 2 w_{bkf}/pool which is characteristic of old-growth streams in the Pacific Northwest (Montgomery et al. 1995).

The pool area equations are less easy to generalize because of their stronger relationship to channel slope. Pool area is more sensitive to LWD abundance in moderate-slope channels ($0.01 < s < 0.04$) than in low-slope channels ($s \leq 0.01$) because numerous free-formed pools compensate for losses of LWD-formed pools in low-slope channels. Therefore, channel width was a better predictor of percent pool in low-slope channels, and LWD abundance was a suitable predictor of percent pool in moderate-slope channels (Beechie and Sibley 1997). Beechie et al. (1994) summarized data from two studies to estimate that percent pool in old-growth channels averages about 65% in low-slope channels, about 45% in moderate-slope channels, and about 35% in steeper channels. Because there were few data in the studies used, the variation about these values was poorly understood. However, I estimate from data in Beechie and Sibley (1997) that percent pool values more than 10% below the average are outside the range of natural variability for streams in the Pacific Northwest.

Duration of effects of altered LWD recruitment on channel morphology – reach scale

If there is no removal of LWD from a stream during logging, the impact of logging on LWD abundance at the time of logging should be negligible. However, depletion of in-channel LWD continues during the period of zero recruitment, leading to a decline in LWD abundance for several decades after a disturbance (Murphy and Koski 1989). Assuming that there is no time lag between loss of LWD and loss of pool formation by LWD, pool abundance will decrease over the same time period as LWD. Recovery of pool formation is a function of recruitment of LWD of sufficient size to form pools, which is a function of stream size (Beechie and Sibley 1997).

Hence, recovery of LWD recruitment is tied to a threshold diameter of LWD. The threshold diameter can be estimated by a regression of the diameter of smallest LWD forming a pool against bankfull channel width, as shown in Figure 5-4 (adapted from Beechie and Sibley 1997). Estimating recovery time for LWD recruitment is then a matter of estimating the time required for trees in the riparian forest to attain sufficient size and begin recruiting to the channel.

This section describes rates of recovery for pool formation by two pathways: the red alder pathway and the Douglas fir pathway. The recovery rate of pool formation by each pathway depends on stand development, LWD input and depletion rates, and channel width and slope. These elements are included in two computer models that are used to describe LWD recruitment over time. This section begins with a description of these two computer models, and then uses these models to describe differences in recovery of LWD recruitment and pool formation as a function of stand type, channel width, and channel slope.

For the purposes of this dissertation, I assume that LWD recruitment immediately after a disturbance is zero, and that LWD recruitment effectively remains zero until trees of sufficient size to form pools are present in the riparian forest. These assumptions are accurate for clearcut logging scenarios but not for fire scenarios where standing dead trees are left near the stream. However, there are no fire-disturbed riparian forests in the study area at present, and the assumptions will not affect the accuracy of recovery time estimates for logging-disturbed riparian forests.

Modeling stand development, LWD recruitment, and pool formation through time

Estimating LWD abundance and pool characteristics through time requires modeling riparian stand development, LWD recruitment from the changing stand, LWD depletion from the channel, and the effect of changes in LWD abundance on pool-riffle morphology. I use two separate computer models to accomplish this (Figure 5-5). The first model, Forest Vegetation Simulator (USDA Forest Service 1996b), simulates growth of a riparian stand given initial plot data. The results of the growth model then become the input data for a second model, Riparian-in-a-Box II (unpublished), which predicts LWD recruitment and changes in LWD abundance and pool-riffle morphology.

The Pacific Northwest Coast variant of the model Forest Vegetation Simulator (FVS) simulates stand growth for a wide range of species and management prescriptions, and produces

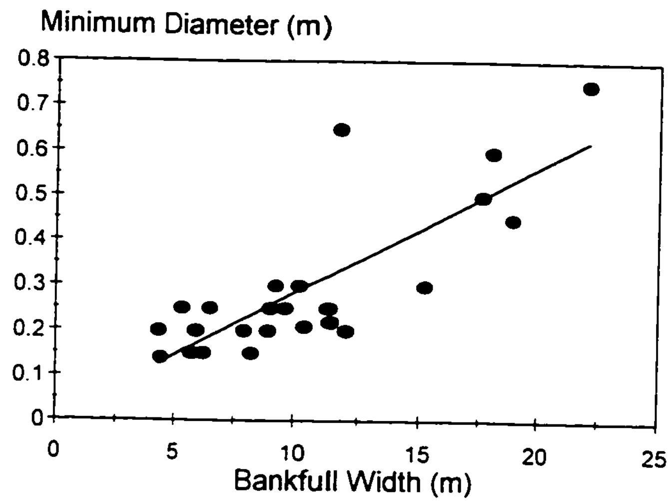


Figure 5-4. Relationship between diameter of smallest LWD formed a pool (diameter_{pf}) and bankfull channel width (w_{bkf}). The regression equation is $\text{diameter}_{pf} = 0.028(w_{bkf}) + 0.0057$ ($r^2 = 0.65$, $P < 0.001$).

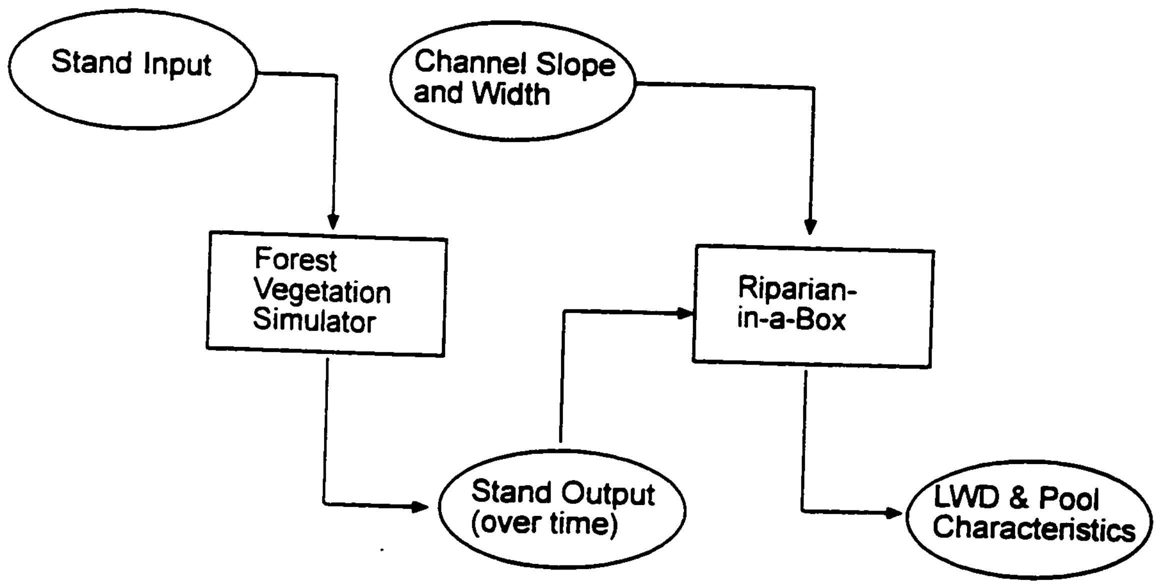


Figure 5-5. Schematic diagram of inputs and outputs for Forest Vegetation Simulator and Riparian-in-a-Box II. Forest Vegetation Simulator models growth of a riparian stand when initial plot data are entered. The results of the growth model then become the input data for Riparian-in-a-Box II, which predicts LWD recruitment, LWD depletion, and changes in LWD abundance or pool-riffle morphology. Channel slope, bankfull channel width, and initial LWD load must also be entered into Riparian-in-a-Box II.

a variety of outputs including basal area and timber volume, diameter and height growth, and trees per acre. For this study I use the model to describe changes in riparian stand characteristics over time and to estimate the number and size of trees that die and can provide LWD to a stream. The most common riparian species in the study area are Douglas fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), western red cedar (*Thuja plicata*), and red alder (*Alnus rubra*). Black cottonwood (*Populus trichocarpa*) and bigleaf maple (*Acer macrophyllum*) are also found in the study area, but are less common in the situations modeled here. For each of the four most common species, I calibrated the model to growth data from local riparian forests (Douglas fir, western hemlock, and red alder) or to growth data from the literature (red cedar). Diameter and height growth were calibrated first for each species, and mortality was calibrated after diameter and height growth were approximately correct. Details of the model calibration and use are in Appendix B.

Kennard et al. (In press) developed a Microsoft® Excel model called Riparian-in-a-Box for estimating rates of LWD recruitment to stream channels as a function of riparian forest management. Its intent was to serve as a management tool for assessing the effectiveness of different riparian management options in terms of LWD abundance. The model described here (Riparian-in-a-Box II) extends the capability of the original, primarily with the addition of equations for estimating the spacing and area of pools based on channel slope, channel width, and LWD loading. Riparian-in-a-Box II (RIAB), has also been recoded as a stand-alone program operating in Windows 95. RIAB first determines which of the fallen trees (from the FVS output) enter the stream by the method of Van Sickle and Gregory (1990). The model then determines the size of LWD entering the stream using simple geometric relationships. Depletion of in-channel LWD is set at a constant value of 1.5% per year in the model, based on Murphy and Koski (1989). LWD abundance is calculated for each time step using the simple equation

$$LWD_{t+1} = LWD_t + recruitment - depletion.$$

Empirical equations then calculate pool area and pool spacing for each reach. Comparisons between scenarios can be based on a variety of model outputs including riparian stand characteristics, LWD abundance, or pool abundance. Details of the model elements and use of the model are described in Appendix B.

With the model structure shown in Figure 5-5, one can ask a variety of questions about the recovery of LWD recruitment and pool formation after a disturbance. With respect to recovery of LWD abundance and pool formation across the landscape, the two main questions are:
(1) How does a single channel type respond to different riparian recovery pathways?
(2) How do channels of different slope respond to the same riparian forest recovery pathway?

The first question targets similar stream types that are bordered by different forest types. However, recovery of LWD recruitment or pool formation cannot be considered independently of channel width because the size of LWD required to form pools varies with channel width. Therefore, estimated time to first recruitment of pool-forming LWD and recovery rate for pool formation each must be considered a function of riparian forest type and channel width. The second question targets situations where similar forest conditions border a variety of stream channel types, which are primarily a function of channel slope. The following three sections describe (1) the effect of riparian recovery pathway and channel width on time to first recruitment of pool-forming LWD, (2) the effect of riparian recovery pathway and channel width on LWD abundance and pool characteristics over time, and (3) the influence of channel slope on LWD abundance and pool characteristics over time.

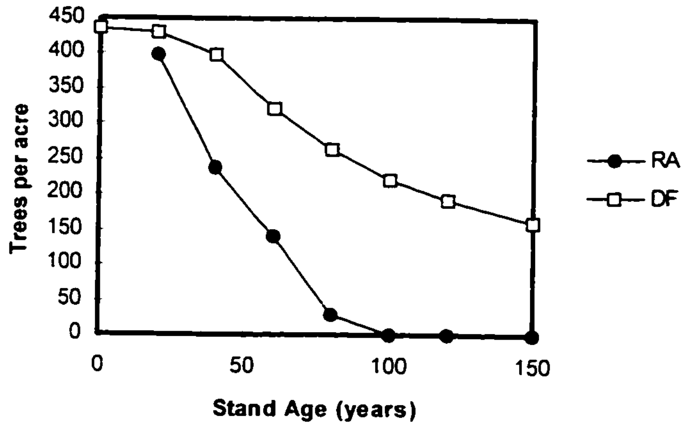
Time to first recruitment of pool-forming LWD by red alder and Douglas fir pathways

Time to first recruitment of pool-forming LWD is governed by size of the channel and growth of the riparian forest. Size of pool-forming LWD is a function of channel width, and growth of the riparian forest is a function of species and site quality. This section first describes stand development for red alder and Douglas fir in terms of trees per acre, diameter, and height. It then describes how time to first recruitment of pool-forming LWD varies for each species and by channel width. In order to focus the modeling on variations in LWD recruitment by species and channel width, I assume that bank erosion is not a dominant recruitment mechanism. The possible consequences of this assumption are addressed in Chapter 6.

The dominant riparian forest types in the study area at present are red alder with little or no conifer in the stand, and Douglas fir plantations along smaller streams that were not buffered historically. Each species has a typical growth pattern (diameter, height, and trees per acre through time), which determines the availability of trees in the riparian forest for LWD recruitment. Red alder typically have relatively high growth and mortality rates in the first few decades, resulting in fewer trees per acre and larger trees up to about age 40 (Figure 5-6). By age 70, alders begin to die off rapidly and relatively few trees are left in the stand by age 100. Douglas fir mortality is lower than that of alder throughout the 150-year modeling time period, so number of stems per acre is always significantly higher than that of an alder stand. Douglas fir height growth begins to exceed that of alder at about age 40 when alder heights are nearing their maximum of 100 to 110 feet (30.5 to 33.5 m). These growth patterns imply that red alder stands have a higher recruitment rate and produce larger LWD during the first few decades, whereas Douglas fir recruitment exceeds that of alder in later decades and persists long after the alder stand has died out.

For channels of similar size, the time lag between a stand replacing disturbance and the first recruitment of pool-forming LWD is a function of tree growth rates in the stand, which is a function of site quality and species. Riparian forest data from the study area indicate that the vast majority of riparian sites are Site Class II (where dominant and co-dominant trees reach an average height of 170 feet at age 100 years), with relatively few sites of Class I (200 feet at age 100 years) or III (140 feet at age 100 years) and virtually no sites of Class IV or V (110 and 80 feet at age 100 years, respectively) (Jan Henderson, plant ecologist, USFS Mount Baker Snoqualmie National Forest, Mountlake Terrace, unpublished data). Therefore, all FVS model runs were based on site quality equivalent to Douglas fir Site Class II, and differences in LWD recruitment due to site quality were not considered. Differences in LWD recruitment due to species differences were evaluated by modeling stands of red alder and Douglas fir, which are the most common hardwood and conifer species, respectively, in the study area. (See Appendix B for further discussion of site quality.)

For this analysis, first recruitment of pool-forming LWD is considered to occur when the quadratic mean diameter of the stand reaches the pool forming diameter for a given channel width. Because alder grows more rapidly than Douglas fir during the first few decades after disturbance, time to first recruitment of pool-forming LWD is shorter for red alder than for Douglas fir at all channel widths (Table 5-4). Based on the modeling results, time to first recruitment of red alder ranges from less than 10 years in small channels to about 50 years in larger channels, and is typically about half that of Douglas fir at all channel widths.



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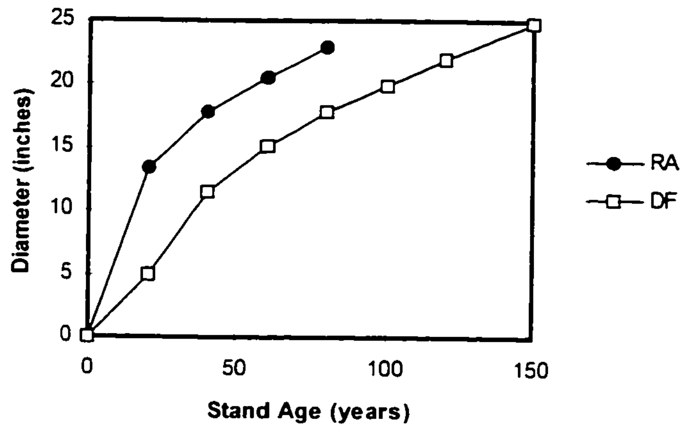

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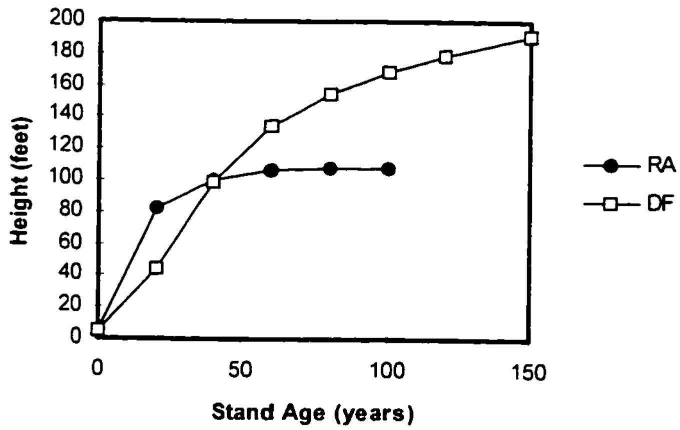


C

Figure 5-6. Calibrated FVS model estimates of trees per acre (A), quadratic mean diameter (B), and height (C) by age for Douglas fir (DF) and red alder (RA) stands.

Table 5-4. Model estimates of time from stand establishment to first recruitment of LWD of pool-forming size, and estimated time to first increase in LWD abundance (in parentheses). The latter does not occur until the recruitment rate for pool-forming LWD exceeds the depletion rate. Pool-forming size of LWD is estimated by the equation: $\text{diameter}_{pf} = 0.025(w_{bkf})$. Channel slope does not affect diameter of pool-forming LWD. NA for alder scenarios means that recruitment does not exceed depletion during the first 100 years, and there are too few trees remaining after 100 years to exceed depletion.

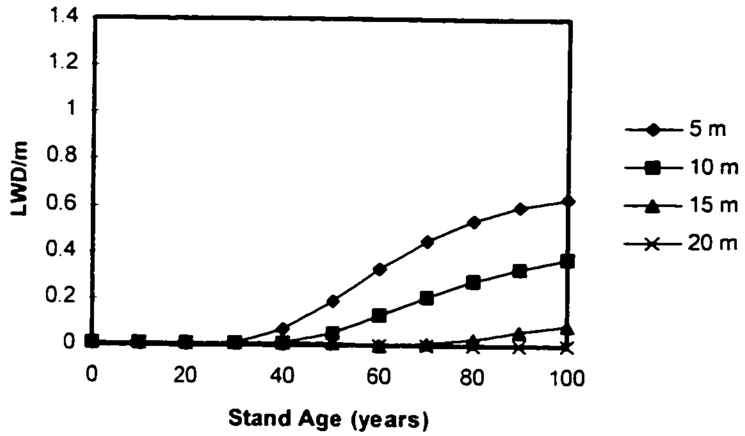
Channel characteristics	Time to recruitment of pool-forming LWD (years) Alder regeneration	Fir regeneration
All slopes ≤ 0.04		
w_{bkf} = 5 m (diameter_{pf} = 13 cm, or 5 in)	7 (10)	15 (30)
w_{bkf} = 10 m (diameter_{pf} = 25 cm, or 10 in)	11 (20)	26 (40)
w_{bkf} = 15 m (diameter_{pf} = 38 cm, or 15 in)	16 (40)	36 (60)
w_{bkf} = 20 m (diameter_{pf} = 50 cm, or 20 in)	21 (NA)	47 (>100)
w_{bkf} = 25 m (diameter_{pf} = 63 cm, or 25 in)	32 (NA)	66 (>100)
w_{bkf} = 30 m (diameter_{pf} = 75 cm, or 30 in)	49 (NA)	91 (>100)

Recovery of LWD abundance and pool characteristics through time

When average diameter at breast height just reaches the diameter of pool-forming LWD, LWD of pool-forming size is recruited only from very near the stream bank. Trees farther away cannot provide large enough LWD to the channel simply because tree diameter decreases with increasing height above breast height, and the portion of a fallen tree that reaches the stream is smaller than the pool-forming diameter. Therefore, the LWD recruitment rate at first recruitment is too low to overcome the depletion rate of LWD, and abundance of LWD in the channel does not begin to increase until somewhat later than first recruitment. LWD abundance first begins to increase when the total recruitment of pool-forming LWD overcomes the depletion, which does not occur until the diameter of LWD originating some distance from the stream is large enough to form pools. Based on the FVS and RIAB models, the time at which LWD recruitment overcomes depletion is typically 10 to 20 years later than the time of first recruitment, and the time lag increases with increasing stream size (Table 5-4).

Once LWD is delivered to a stream, the ultimate abundance and size distribution of LWD are governed primarily by channel size. As channel size increases, LWD abundance decreases and average LWD size increases (Bilby and Ward 1989). These changes are primarily attributed to the increased mobility and depletion of smaller LWD as channel size increases, which results in fewer and larger pieces left in larger channels. Pool formation by LWD is influenced both by channel size and by channel slope. The size of LWD required to form a pool increases with increasing channel width, which is at least partly related to the decreasing stability of smaller LWD with increasing channel width (Beechie and Sibley 1997). Channel slope primarily affects the sensitivity of pool spacing and pool area to changes in LWD abundance (Beechie and Sibley 1997). At similar LWD abundance, lower slope channels have more pools and higher percent pool area than do steeper channels, which is primarily a result of the increased abundance of free-formed pools in lower slope channels (Montgomery et al. 1995, Beechie and Sibley 1997).

For the red alder and Douglas fir pathways LWD abundance begins to increase sooner at smaller channel widths, and LWD abundance remains higher for smaller channels than for larger channels in all subsequent years (Figure 5-7). Neither pathway produces sufficient pool-forming LWD to overcome the depletion rate during the first 100 years after disturbance for any channel width greater than 20 m. For channels <10 m wide, LWD abundance is higher for the alder recovery pathway than for the Douglas fir pathway during the first 100 years. However, Douglas



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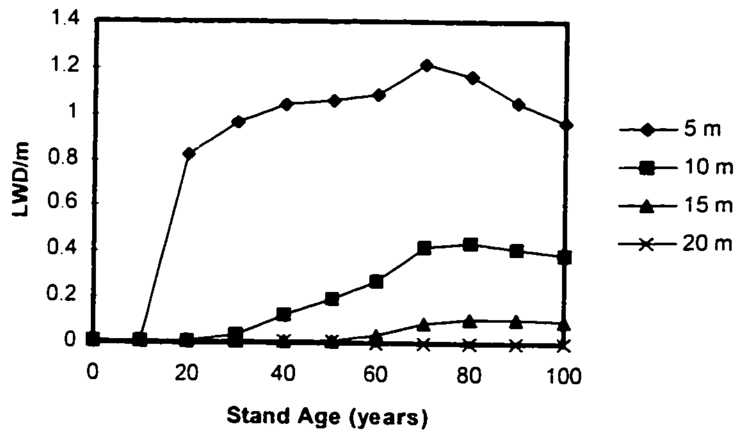

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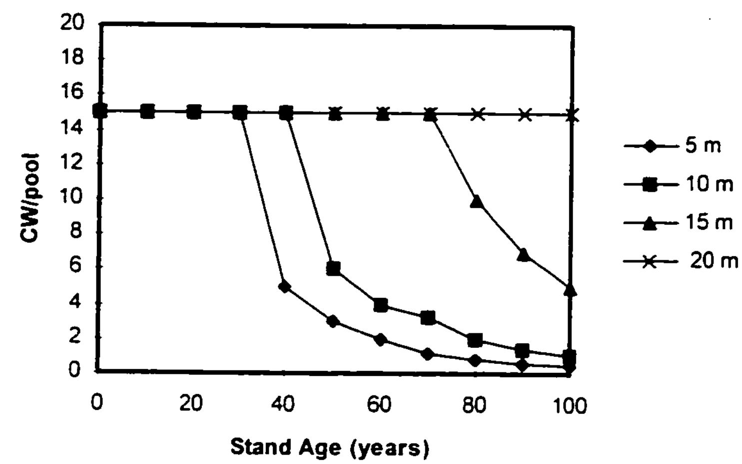
Figure 5-7. LWD abundance (LWD/m) in channels of different widths (listed in legend) for (A) the Douglas fir recovery pathway and (B) the red alder pathway. For all channels ≥20 m, LWD recruitment does not exceed depletion and LWD/m declines throughout the 100 year modeling period.

fir LWD abundance in channels ≤ 15 m wide should overcome alder LWD abundance after 100 or more years (Andrus et al. 1988). For channels ≥ 20 m wide, Douglas fir abundance should begin to increase more than 100 years after disturbance, and should exceed that of alder in all subsequent years.

Pool spacing and percent pool area are primarily functions of LWD abundance and channel slope (Beechie and Sibley 1997), and LWD abundance is strongly related to channel width (Bilby and Ward 1989). Thus, for a given riparian recovery pathway, recovery of pool formation in different channel types is expected to vary with channel width and slope. Based on the argument that smaller LWD forms pools in smaller channels (Beechie and Sibley 1997), one expects that pool abundance would increase most rapidly in smaller channels. FVS and RIAB model results support this prediction for both red alder and Douglas fir recovery pathways. Using the Douglas fir recovery pathway to illustrate changes in pool formation as a function of channel width, the model predicts that pool spacing declines most rapidly in small channels, and that percent pool increases most rapidly in small channels (Figure 5-8). Because LWD recruitment does not exceed depletion for channel widths ≥ 20 m, number of pools and percent pool continually decrease during the 100-year simulation. (Simulations are restricted to 100 years because confidence in FVS outputs is low for older stands.)

Effect of channel slope on recovery of pool characteristics

Channel slope has its strongest effect on pool abundance at low LWD abundance because free-formed pools are more common in low slope channels. To illustrate the effects that channel slope has on pool formation, I modeled the Douglas fir recovery pathway at a channel width of 10 m for several channel slopes. As expected, the model predicts greater changes in pool abundance in steeper channels where free-formed pools are rarely formed (Figure 5-9). In channels with slope between 0.01 and 0.04, predicted pool spacing is extremely wide during the first few decades when LWD abundance is low and decreases in later decades as LWD abundance increases. Pool area increases with increasing LWD abundance for channels within this slope range, but pool area increases with decreasing channel slope. For channels with slope less than 0.01, the model predicts a pool spacing of 6 channel widths per year at low LWD abundance (based on Leopold et al. 1964), and decreasing pool spacing at higher LWD abundance. Pool area is not responsive to LWD abundance when slope is less than 0.01.



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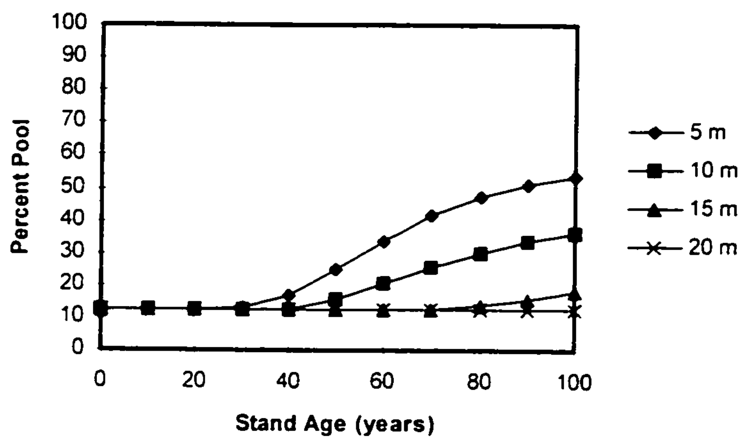

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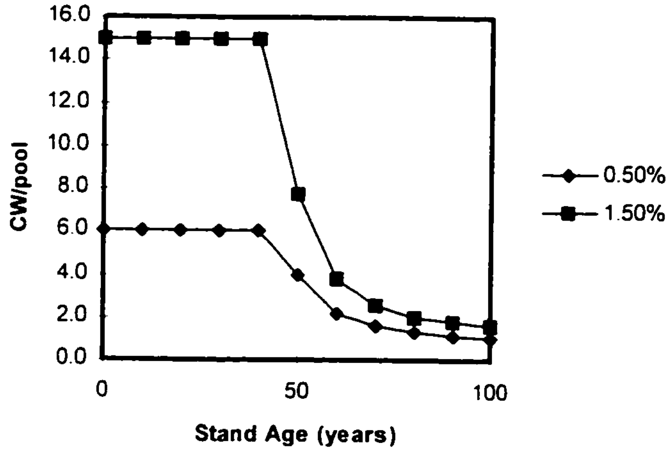
Figure 5-8. Simulated variation in recovery of (A) pool spacing (CW/pool on the graph) and (B) percent pool as a function of changes in channel width (listed in legend). Simulations are based on the Douglas fir recovery pathway. Channel slope for this simulation is 3.0%.



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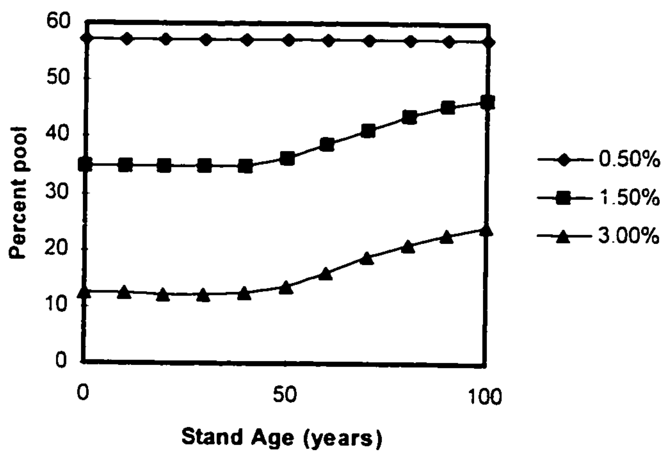

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Figure 5-9. Simulated variation in recovery of (A) pool spacing and (B) percent pool in relation to channel slope (listed in legend). Simulations are based on the Douglas fir recovery pathway. Channel width for these simulations is 10 m. Note that predicted pool spacing is insensitive to changes in LWD abundance when LWD/m <0.05. Pool spacing is set at 6 w_{bkf} /pool (CW/pool on the graph) for channels <0.01 slope based on the pool spacing of 5 to 7 channel widths found in low slope channels without LWD (Leopold et al. 1964). Pool spacing is set at 15 w_{bkf} /pool for channels with slope between 0.01 and 0.04 based on the widest pool spacing measured by Montgomery et al. (1995).

Landscape-level patterns of channel response to altered LWD recruitment

Many studies in the Pacific Northwest show that logging near streams causes decreased LWD abundance in channels (e.g., Murphy and Koski 1989, Carlson et al. 1990, Bilby and Ward 1991, Ralph et al. 1994). However, no one study describes a region-wide decrease in LWD abundance, although it is evident from these studies and a number of watershed analyses (e.g., WDNR 1994) that most reaches across the landscape have reduced LWD abundance. Other studies indicate that the decrease in LWD abundance affects different types of stream channels in different ways (e.g., Bilby and Ward 1989, Beechie and Sibley 1997, Montgomery and Buffington 1997). However, they generally do not describe patterns of LWD reduction and its effects on channel morphology and habitat across the landscape.

LWD functions vary systematically with location in the channel network. In steeper channels (mostly >0.04), LWD does not appreciably influence pool spacing (Montgomery et al. 1995), but has an influence on pool size and sediment storage (Megahan 1982, Bilby and Ward 1989). There is also evidence that valley spanning LWD jams can force steep bedrock channels to alluvial channels by reducing the local channel slope and storing sediment (Montgomery et al. 1996). Key members of LWD in steeper channels may not be systematically related to channel size in the same way that they are in lower-slope channel, but diameters of key LWD in these steeper channels typically exceed 0.6 m (Montgomery et al. 1996).

As described earlier, pool formation in low-slope (<0.01) and moderate-slope (0.01-0.04) channels is strongly influenced by LWD abundance. Pool formation is more sensitive to LWD abundance in moderate-slope channels, but LWD abundance remains a strong influence on pool spacing even in low-slope channels (Beechie and Sibley 1997). The minimum size of pool forming LWD in a reach is a function of channel size, which in part governs the rate at which pool formation can recover after a decrease in LWD abundance (Beechie and Sibley 1997).

Declines in LWD and pool abundance are driven primarily by depletion of LWD during period of low LWD recruitment after disturbance to a riparian forest. Depletion of smaller LWD occurs faster than larger and more stable LWD (Murphy and Koski 1989). Furthermore, the size of the smallest LWD that remains stable increases with increasing channel size (Bilby and Ward 1989). These two facts suggest that LWD should deplete faster on average in larger channels because a greater proportion of the recruited LWD (i.e., smaller pieces) is more easily transported.

In 1st and 2nd order streams with steep slopes (>0.04), channels are typically narrow and smaller LWD is stable in these channels (Bilby an Ward 1989). However, key pieces that initiate formation of step-forming jams are much larger (Montgomery et al. 1996), and sediment storage is also greatly increased with increasing LWD size (Bilby and Ward 1989). Thus, LWD >0.6 m diameter is required to maintain a channel morphology that is associated with typical late-seral forest conditions (Montgomery et al. 1996). Time to first recruitment of stable LWD in these channels is approximately 100 years (Table 5-5), and the time to recovery of a channel morphology and sediment storage typical of late-seral conditions could take as long as 250 years.

In low-slope and moderate-slope channels the size of LWD required to initiate pool formation is a function of channel width, so recovery time to first recruitment of pool-forming LWD is variable. In smaller 3rd and 4th order channels bankfull widths range from about 4 to 15 m in width and the minimum size of pool-forming LWD ranges from 10 cm to 40 cm. Recovery to first recruitment of pool-forming LWD is about 30 to 60 years in these channels depending on riparian species and channel size (Table 5-5). However, recovery of pool abundance to approximately late-seral conditions will probably take more than 100 years. In 5th and 6th order streams typical bankfull widths range from 15 to 25 m in width and the minimum size of pool-forming LWD ranges from 40 cm to 60 cm. Recovery to first recruitment of pool-forming LWD is about 60 to 100 years in these channels, and recovery to late-seral conditions is more than 150 years. In larger streams (>7th order, >25 m wide) pool-forming LWD is >70 cm, and recovery to first recruitment of pool-forming LWD will likely exceed 120 years. Recovery of LWD and pool abundance to approximately late-seral levels will take more than 200 years.

The recovery potential of individual stream reaches with respect to LWD is here considered equivalent to the time to first recruitment of pool-forming LWD. In general, smaller streams have higher recovery potential because the time to first recruitment of pool-forming LWD is shorter. Time to first recruitment of pool-forming LWD (or recovery potential) can be mapped based on the time required for a riparian forest to attain an average diameter at breast height equal to the pool-forming diameter. For this example, the Douglas fir recovery pathway is used to estimate recovery time to recruitment of pool-forming LWD, as shown earlier with the FVS modeling of recovery time for channels of different width (Table 5-4). The diameter of pool-forming LWD ($diameter_{pf}$) is estimated as a function of channel width (w_{bkf}):

Table 5-5. Typical duration of reduced LWD and pool abundance in different channel types in western Washington. Channel slope and morphology descriptions based on Montgomery and Buffington (1997). Typical pool forming sizes based on Beechie and Sibley (1997). Recovery to first pool formation based on RIAB II modeling results.

Stream order	**Typical channel slope and morphology**	**Typical pool-forming or stable LWD size**	**Typical recovery time to first pool formation**	**Typical duration of reduced LWD and pool abundance**
1-2	>18%: colluvium, bedrock	>60 cm	>100 yrs	>150 yrs
3-4	8-20%: boulder cascade	>60 cm	>100 yrs	>150 yrs
	3-8%: step pool	20-40 cm	30-60 yrs	>100 yrs
	<4%: pool-riffle, plane-bed, forced pool-riffle	20-40 cm	30-60 yrs	>100 yrs
5-6	8-20%: boulder cascade	>60 cm	>100 yrs	>150 yrs
	3-8%: step pool	40-60 cm	60-100 yrs	>150 yrs
	<3%: pool-riffle, plane-bed, forced pool-riffle	40-60 cm	60-100 yrs	>150 yrs
>7	<1%: pool-riffle, forced pool-riffle	> 70 cm	>120 yrs	>200 yrs

$diameter_{pf} = 0.025(w_{bkf})$,

where $diameter_{pf}$ and w_{bkf} are in meters. (See appendix B for further explanation of the LWD diameter equation.) Channel width can be estimated as a function of drainage area (A):

$w_{bkf} = 4.2(A)^{0.49}$,

where w_{bkf} is in meters and A is in km^2. This equation is based on a regression of channel width on drainage area for unconfined reaches in second-growth forests in the study area ($n = 23$, $r^2 = 0.88$).

In general, smaller low-slope and moderate slope streams have the greatest recovery potential (shortest time to recruitment of pool-forming LWD) and larger and steeper streams have lower recovery potential (longest time to recruitment of pool-forming LWD), as illustrated for a five square mile area of Bacon Creek sub-basin (Figure 5-10). Small tributaries such as Oakes Creek have high recovery potential because smaller LWD can form pools and recruitment of pool-forming LWD begins to increase within about 17 years. By contrast, a larger tributary such as Falls Creek has a lower recovery potential (about 34 years) because larger LWD is required to form pools. The lowest recovery potentials are in even larger channels such as Bacon Creek, where abundance of pool forming LWD does not begin to increase until after at least 100 years.

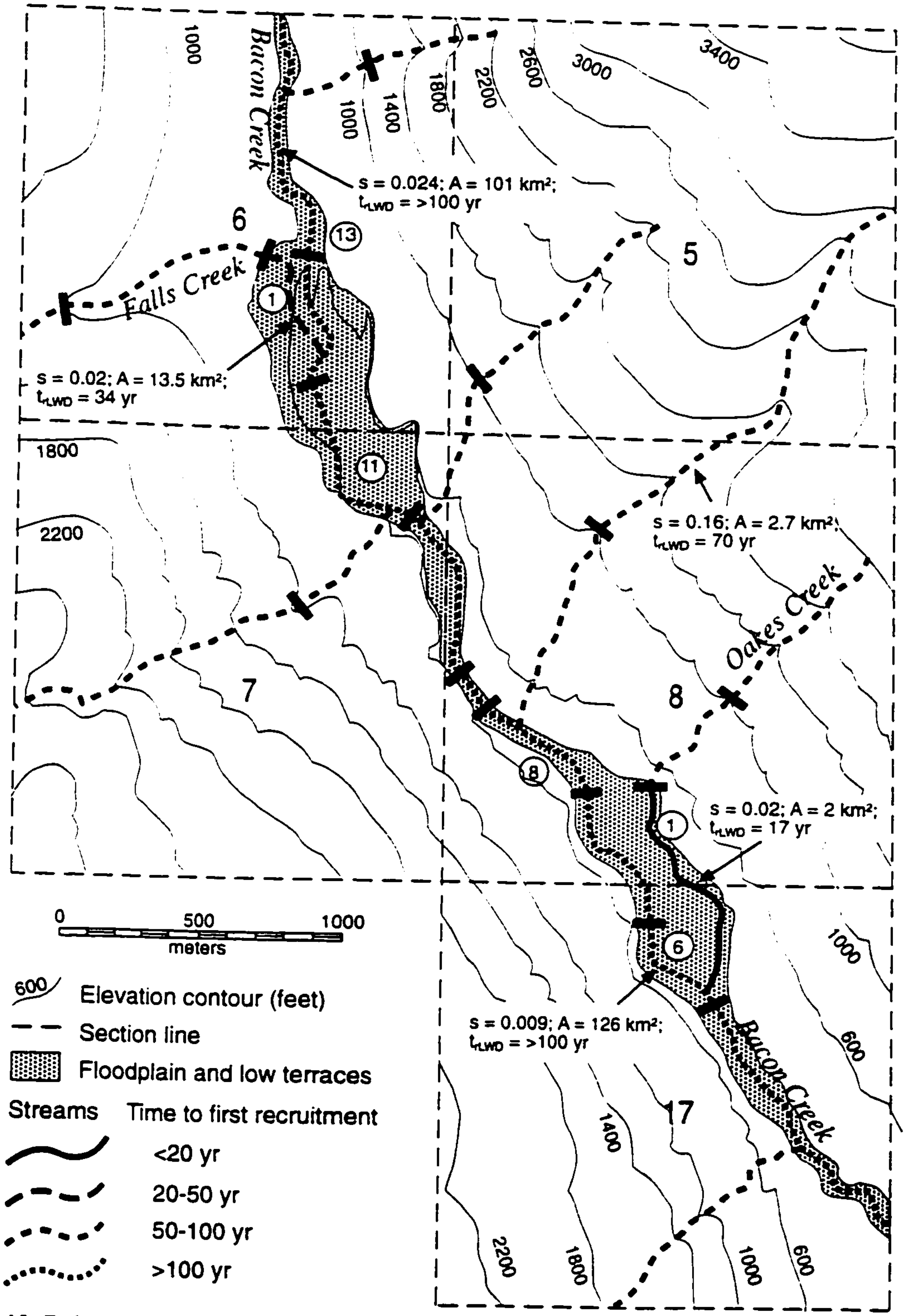


Figure 5-10. Estimated time to first recruitment of pool-forming LWD and relative recovery potentials for a five square mile area of Bacon Creek sub-basin. Solid line indicates greatest recovery potential from loss of LWD (i.e., shortest time) and smallest dashes represent lowest recovery potential (i.e., longest time). Selected reach numbers in circles.

Chapter 6. Application

There are many factors besides freshwater habitat conditions that may inhibit recovery of fish populations (e.g., changes in ocean conditions, fishing pressure, interactions with hatchery stocks), but degradation of habitats is one of the causal factors in virtually all cases of salmonid stock declines in the Pacific Northwest (Nehlsen et al. 1991). Many of these habitats can be expected to remain degraded for long time periods (e.g., Bisson et al. 1997), and fish populations typically will not recover until habitat conditions have recovered (Yount and Niemi 1990). However, once physical and chemical characteristics of habitats have recovered, fish populations generally recover within a few years (Niemi et al. 1990). Because recovery of fish populations is in part limited by the slow recovery of habitat characteristics, it follows that speeding the recovery of habitat conditions should help achieve a more rapid recovery of salmon populations.

Production of salmonid smolts in the study area appears to be limited primarily by isolation and physical alteration of habitats, but also by degradation resulting from altered riparian functions and sediment supply (e.g., Beechie et al. 1994, Collins et al. 1994, Lunetta et al. 1997). Where isolated intact habitats can be reconnected in the study area, recovery of habitat function is rapid and fish populations recover relatively quickly (Beamer et al. 1998). In most cases, problems of isolated habitat require no further research to understand how best to restore them, and reconnection of isolated habitats is now under way in the Skagit River basin. Aside from habitat isolation, most of the remaining physical habitat losses are functions of changes in supply sediment and LWD to streams. Therefore, this dissertation has focused on understanding the rates and pathways of recovery for these two habitat-forming processes.

As defined in chapters 4 and 5, recovery potential is a measure of the capacity of a given stream reach to recover from a disturbance. Recovery potential from sediment supply disturbances is measured as the annual travel distance of sediment, which is a function of channel slope and bankfull discharge. In other words, the sediment supply recovery potential is a measure of the rate at which a reach can transmit or evacuate an increased sediment supply. Recovery potential from LWD recruitment disturbances is the time to first recruitment of pool-forming LWD, which is primarily a function of channel width and growth rates of the riparian forest. It is therefore a measure of time required for trees to grow large enough to create pools when they are recruited to the stream channel.

The intent of describing disturbance and recovery pathways for sediment supply and LWD recruitment (Chapters 4 and 5) was to provide information necessary for sensible restoration planning. In that context, the main uses of the recovery pathways are in estimating the length of time necessary for habitats to recover, and in estimating the effectiveness of alternative restoration options. Estimates of recovery time from current conditions are based on (1) the rate of recovery for a given pathway, and (2) the amount of recovery that has already occurred. The effectiveness of different restoration options depends on (1) how much a specific restoration action can increase the recovery rate for a given pathway, and (2) the amount of recovery that has already occurred. These types of estimates can help narrow the range of options available to restoration planners, and can focus restoration efforts on those areas where habitat improvements are likely to be greatest.

For each recovery pathway described in chapters 4 and 5, I estimated recovery rates as a function of channel slope and size. However, these recovery rates are essentially deterministic and lack consideration of other future disturbances. Recovery pathways and rates for either sediment supply or LWD recruitment can be altered by many processes, including continued or repeated disturbances of the same type (e.g., more landslides delivering sediment to a reach that is recovering), disturbances of a different type (e.g., channel migration or avulsion eliminating a newly colonized riparian forest), or physical conditions of a channel that alter the expected recovery rate (e.g., LWD slowing the rate of sediment transport through a reach). Some of these factors can be considered in restoration planning by estimating their relative probabilities of occurrence in different parts of the channel network. Such considerations can help identify reaches with higher likelihood of future disturbance and failure of restoration projects.

This chapter presents two examples of the use of the recovery pathways in restoration planning. The first example describes the estimation of recovery potential and landscape-level mapping of recovery potentials for disturbances to both sediment supply and LWD recruitment. Having a theoretical basis for describing rates and pathways of recovery allows restoration planners to understand where recovery is likely to be most rapid. Therefore, even in the absence of other data it is possible to map locations where restoration efforts can be most cost-effective. The first example also describes the probability that other processes will alter recovery rates, and maps indicators of these relative probabilities at the landscape level. Together, the mapped recovery potentials and probabilities of future disturbance give a landscape-level view of where certain types of recovery will most rapidly occur, and where the likelihood of successful restoration projects is relatively greater.

The second example illustrates how additional data describing the degree of disruption to a watershed process and the length of time since the disturbance can help planners more accurately gauge the remaining recovery time for a given process. This additional information is essential to making more accurate predictions of where restoration actions are most likely to be cost-effective. This second example focuses on LWD recruitment, showing how recovery time can be estimated based on recovery potential and the present level of disturbance. It then describes procedures for enhancing or speeding the recovery rates of LWD recruitment, and illustrates the use of these methods in delineating where restoration actions are most warranted based on relative degree of improvement in recovery time.

Example 1: Estimating and mapping recovery potentials

As described in preceding chapters, one can classify the relative recovery potentials of individual reaches in the channel network for an increase in sediment supply (Chapter 4) or a decrease in LWD recruitment (Chapter 5). In the simplest terms, sediment supply recovers more quickly in larger or steeper channels because higher sediment transport capacities can more rapidly evacuate stored sediment from a stream reach. By contrast, recruitment of pool-forming LWD recovers more slowly in larger or steeper channels because it takes longer to grow trees large enough to influence pool formation. Thus, in the absence of further analysis, the two recovery potentials for a given stream reach will most commonly suggest different restoration priorities for each process (see Figures 4-9 and 5-10).

Assessment of the probability that other processes may inhibit recovery of either sediment supply or LWD recruitment can help delineate where the original estimates of recovery potential must be used with greater caution. This section presents examples of mapping recovery potentials and probabilities of future disturbance from certain processes. It also discusses the relative importance of sediment supply and LWD recruitment in recovery of salmonid rearing habitats, and suggests that recovery of LWD recruitment most limits recovery of rearing habitats.

Sediment Supply

This section first describes the probability that LWD in stream channels will significantly affect recovery of sediment supply based on a brief literature review. The literature review illustrates relationships between channel size and the relative degree to which LWD can inhibit sediment transport, and describes a simple classification of these effects by channel size. Following this description, a mapping example illustrates the classification of recovery potentials based on channel size (as described in Chapter 4) and demonstrates mapping of the relative probability that LWD in the channel will alter recovery potential.

Several studies indicate that LWD can store sediment (Megahan 1982, Bilby and Ward 1989, O'Connor and Harr 1994) or inhibit sediment transport by locally dissipating stream energy (Buffington 1995). LWD accumulations can function as sediment storage dams (Megahan 1982), which also reduce the local stream slope and sediment transport capacity (O'Connor and Harr 1994, Montgomery et al. 1997). LWD pieces in stream channels also function as large roughness elements that reduce sediment transport capacity (Buffington 1995). These studies indicate that the presence of LWD can reduce the rate of sediment movement through a stream channel, and may therefore lengthen the recovery time from an increase in sediment supply. However, these types of LWD functions typically occur only in smaller channels (<15 m) (Keller and Swanson 1979) (Table 6-1). LWD accumulations in larger channels (>30 m) generally do not span a significant part of the stream channel and therefore do not significantly affect sediment transport (Keller and Swanson 1979, Abbe and Montgomery 1996). LWD in channels of intermediate width (15-30 m) is considered to have an intermediate effect on sediment transport.

LWD may also influence recovery time from increased sediment supply when channel widening occurs as a result of increased sediment supply. In such cases, mature riparian forests may help reduce recovery time by providing LWD as a channel erodes its banks, which can protect bar surfaces from high energy flood flows and enhance recovery of riparian vegetation (Collins et al. 1994, Featherston et al. 1995, Abbe and Montgomery 1996). This process, which occurs mostly in larger channels (>30 m), increases the rate at which sediment is transferred from in-channel storage to floodplain storage (Featherston et al. 1995). Under logged conditions, channel widening does not increase LWD recruitment because the trees have been removed, and part of the natural recovery process is eliminated (Collins et al. 1994). Without stable debris pieces or jams the channel is more prone to lateral migration, which continually reactivates channel-stored sediment and inhibits the recovery of sediment supply.

Mature riparian vegetation may also inhibit channel widening, which decreases the amount of sediment recruited to the channel from floodplain storage and decreases the area of denuded floodplain. Reduced channel widening and migration may decrease recovery time by decreasing the quantity of sediment that must be transferred out of the reach and by eliminating the time required to re-establish floodplain vegetation. Larger riparian vegetation will also inhibit lateral migration of channels, which also will help decrease the amount of sediment recruited to the channel from the floodplain.

As described in Chapter 4, mapping of reaches with different relative recovery potentials can help planners understand where recovery is likely to be most rapid, or where recovery will be slower. These maps are useful where different sediment supply reduction efforts are being compared for cost effectiveness, or where knowledge of the recovery time for sediment supply is integral to the planning of other restoration projects (e.g., Collins et al. 1994). For this example, I mapped reaches with four different levels of recovery potential based on channel slope and bankfull channel width (w_{bkf}), which can be related to annual travel distance (L_b) of sediment by the equation $L_b \cong 20(w_{bkf})$ (see Figure 4-8). Recovery potentials for different channels are first classified by slope, with slope >0.04 having high recovery potential but low potential to benefit the salmonid resource. These steep channels typically do not support anadromous fish populations (Lunetta et al. 1997). Channels with slope less than 0.04 have varying recovery potentials as a function of stream size. Using the channel width categories described above, I classified channels with slope <0.04 into categories of <15 m wide, 15-30 m wide, or >30 m wide, which correspond to approximate annual travel distances of <300 m/yr, 300-600 m/yr, and >600 m/yr (Table 6-1). For this example, these classifications are mapped in the 140 km^2 Finney Creek and 30 km^2 Hansen Creek sub-basins (Figures 6-1 and 6-2).

Where channels have widened significantly due to extreme floods or sediment supplies, channel width may not be a reliable indicator of sediment transport rates, and one can substitute drainage area for stream size within the study area. Channel width is related to drainage area within the study area by the equation $w_{bkf} = 4.2(A)^{0.49}$ (Figure 6-3). Drainage areas of channels less than 15 m wide are typically less than 15 km^2 in the study area, and drainage areas of channels more than 30 m wide are typically greater than 55 km^2. Within the study area, mapping

Table 6-1. Variation in annual travel distance and likelihood of slowed recovery time due to LWD loading.

Channel width	Annual travel distance	Likelihood of slowed recovery due to LWD loading
< 15 m	<300 m/yr	High
15-30 m	300-600 m/yr	Moderate
> 30 m	>600 m/yr	Low

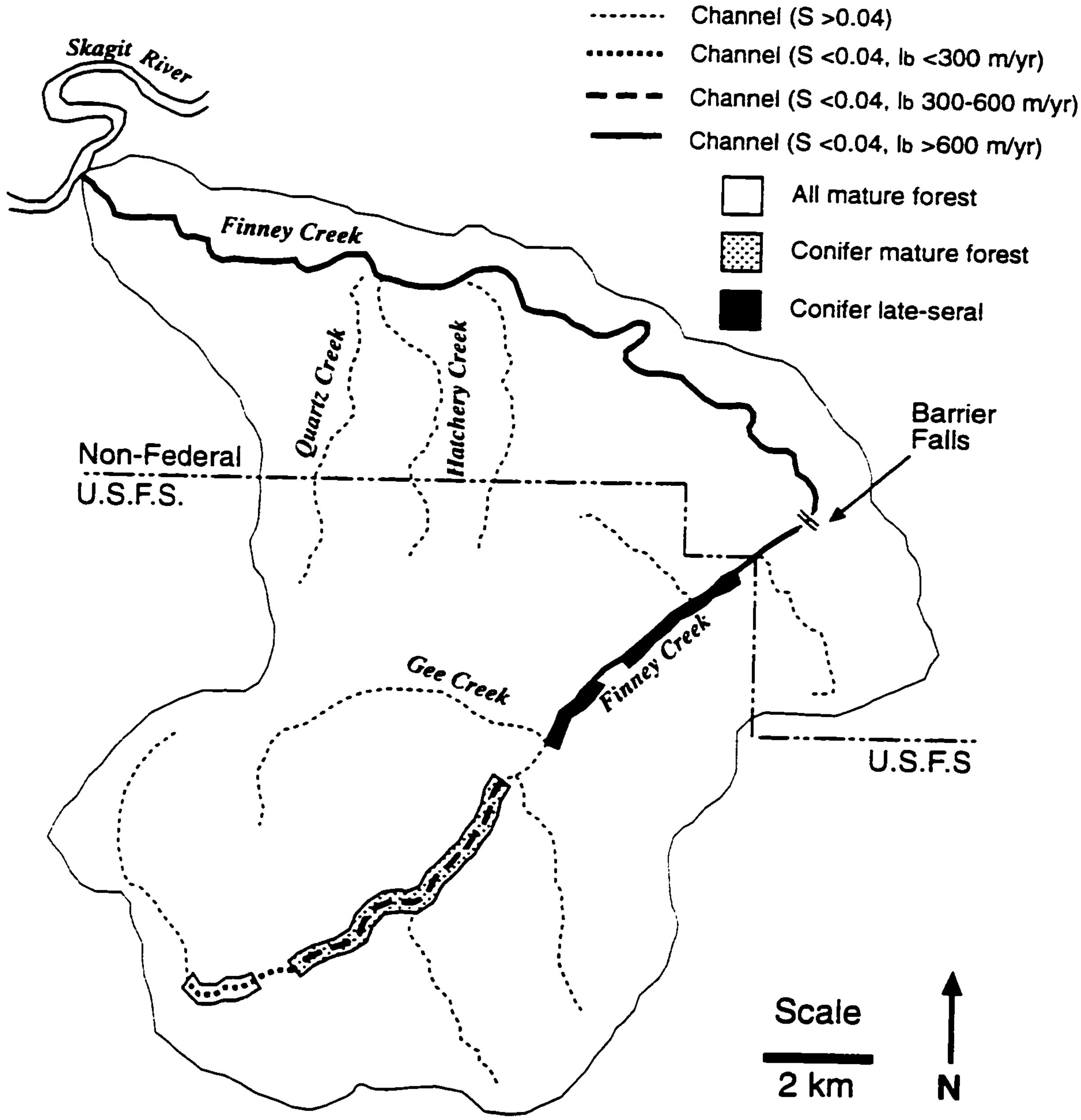


Figure 6-1. Sediment supply recovery potentials (expressed as annual travel distance in meters per year) for the 140 km^2 Finney Creek sub-basin in Skagit County, Washington. "All mature forest" for channels less than 15 m wide includes all hardwood and conifer forest types. "Conifer mature forest" for channels 15 to 30 m wide includes only mid-seral and late-seral conifer forests. "Conifer late-seral" forests mapped along channels greater than 30 m wide include only late-seral conifer forests. All forest types are based on 1991 and 1993 Landsat data, as mapped by Lunetta et al. (1997).

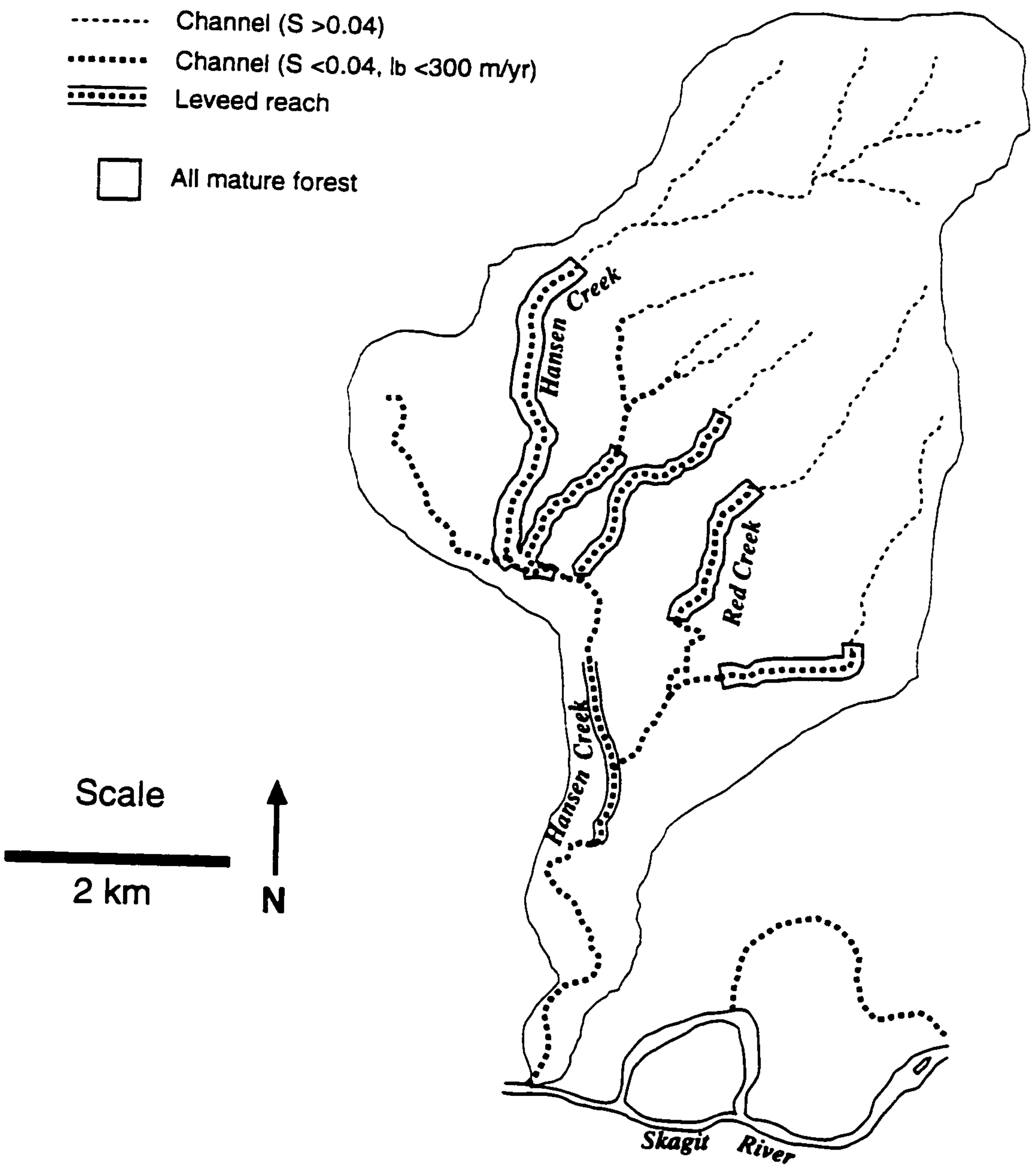


Figure 6-2. Sediment supply recovery potentials (expressed as annual travel distance in meters per year) for the 30 km^2 Hansen Creek sub-basin in Skagit County, Washington. "All mature forest" for channels less than 15 m wide includes all hardwood and conifer forest types. Forest types are based on 1991 and 1993 Landsat data, as mapped by Lunetta et al. (1997).

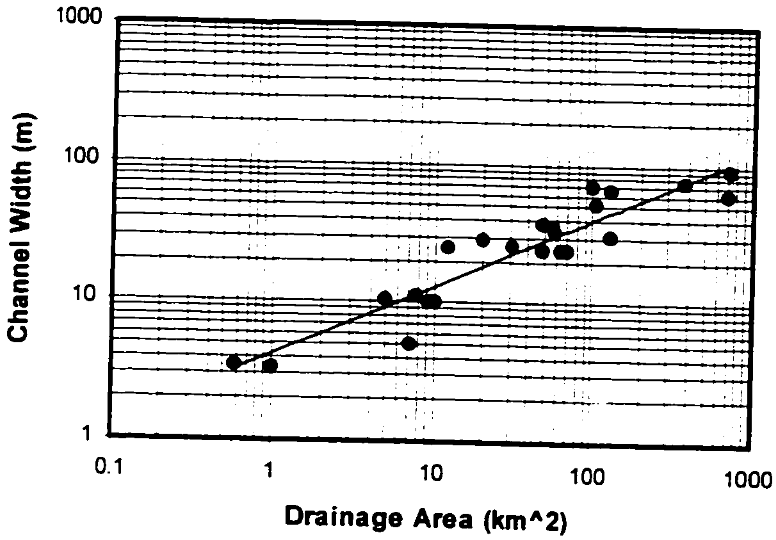


Figure 6-3. Bankfull channel width (w_{bkf}) as a function of drainage area (A) in channels with forested riparian zones. Regression equation is $w_{bkf} = 4.2(A)^{0.49}$ ($r^2 = 0.87$). Data are from forest channels in the Stillaguamish River basin (Beechie et al. 1997).

of these drainage area classes will produce results that are approximately equivalent to those illustrated in Figures 6-1 and 6-2.

In addition to mapping recovery potentials, one can also estimate whether LWD loading may inhibit sediment transport, thereby giving additional information to restoration planners. As described earlier, channels <15 m wide not only have lower recovery potential due to lower sediment transport capacity, but also a higher likelihood of slowed recovery time due to the effects of LWD loading (Table 6-1). High LWD loading in these channels creates a forced pool-riffle channel morphology (Montgomery et al. 1995), which is associated with increased channel roughness and reduced sediment transport (Buffington 1995). Because high LWD loading is found in approximately 75% of reaches with forested riparian areas (Lunetta et al. 1997), we can map riparian forest conditions along channels less than 15 m wide to indicate where LWD accumulations are most likely to inhibit sediment transport (Figures 6-1 and 6-2).

Although reductions of sediment transport rates are less likely in channels between 15 m and 30 m wide, mapping of mature conifer forests along these reaches also gives planners an indication of where reduced recovery potential due to high LWD loading may occur. Larger LWD is required to significantly affect channel morphology in these mid-size streams (0.4 m to 0.8 m diameter LWD based on Beechie and Sibley 1997), indicating that red alder forests are unlikely to provide suitable LWD. Thus only mid-seral and late-seral conifer forests are included on the maps for mid-size reaches (Figure 6-1).

It is unlikely that LWD will significantly affect sediment transport rates and reduce recovery potential in channels greater than 30 m wide. However, channels greater than 30 m wide are more likely to have increased recovery potential where LWD helps increase the rate of revegetation on denuded bars and floodplain (Featherston et al. 1995, Abbe and Montgomery 1996). These channels typically require LWD with a mean diameter of at least 0.8 times the average depth of the channel (Abbe et al. 1997), and channel depths often exceed 1 m (Abbe and Montgomery 1996). Thus, only late-seral conifer forests are likely to provide significant amounts of sufficiently large LWD to these larger channels. For the purpose of this example, I mapped only late-seral forests along channels wider than 30 m (Figure 6-1).

LWD Recruitment

This section first describes the probability that channel migrations or avulsion will significantly affect recovery of LWD recruitment based on a brief literature review. The literature review illustrates relationships between channel size and the relative probability of channel migration or avulsion in general terms, and describes a simple classification of these probabilities relative to channel size. Following this description, a mapping example illustrates the classification of recovery potentials based on channel size (as described in Chapter 5) and demonstrates mapping of the relative probability that channel migration or avulsion will alter the recovery potential.

Most channel movements occur during floods, although increased sediment supply may also cause or exacerbate widening or migration of unconfined stream channels (see Chapter 4). Unconfined channels >30 m wide tend to have high lateral migration rates, even when bordered by old growth forests (Featherston et al. 1995, Abbe and Montgomery 1996). These channels have wide floodplains, and channel migration is a nearly annual occurrence in the largest channels. Scroll bar formation on very large channels (>75 m wide) indicate that lateral migration is a frequent occurrence on channels with sufficient stream power to erode banks despite the root strength of mature forests (Figure 7 in Featherston et al. 1995, Figure 2 in Abbe and Montgomery 1996). However, in smaller channels (about 30 m wide) vegetation patterns suggest less frequent channel migration or widening and indicate that bank erosion occurs only during years of larger floods (Collins et al. 1994, Abbe and Montgomery 1996). Collins et al. (1994) found that reaches as large as 40 m bankfull width may sometimes be stable over several decades under old-growth conditions, but begin widening or migrating after riparian logging. Where riparian forests have been logged, channels become more susceptible to widening or migration, primarily as a result of reduced root reinforcement of banks (Collin et al. 1994). In combination, these observations suggest that channels tend to exhibit high lateral migration rates under all forest conditions when bankfull channel width exceeds 40 or 50 m. However, channels less than about 30 m wide will tend to exhibit significant lateral migration or widening only under immature forest conditions.

Where channel migration rates are high, riparian forests are frequently disturbed and rarely reach the age of mature conifer forests (Abbe and Montgomery 1996). Moreover, frequently shifting channels and erosion of bar surfaces may inhibit colonization, thereby prolonging the recovery of riparian forests and LWD recruitment. Because lateral migration is typical of larger channels, the probability that channel migration will hinder vegetation recovery is considered high in relatively unconfined channels that are wider than 30 m (drainage area >55 km^2, and with a relatively wide flood plain). Where large channels are confined by valley walls, channel migration or widening will not significantly affect the recovery of LWD recruitment (Table 6-2).

Lateral migration and channel widening are less likely in smaller channels, especially in those less than 15 m wide. Although some bank erosion and migration occur even in small channels, significant alteration of riparian vegetation is rare because rates of bank erosion are negligible. For example, even with very high sediment supply in the early 1990's, the 8 to 11 m wide Hansen Creek channel did not widen enough to affect riparian vegetation (WDNR 1994). Channels <15 m wide thus have very low probability that channel migration or widening will affect recovery of riparian vegetation and LWD recruitment (Table 6-2). Channel widening and migration in channels of intermediate width (15 to 30 m) has a moderate likelihood of affecting riparian vegetation.

As described in Chapter 5, mapping of reaches with different relative recovery potentials can help planners understand where LWD recovery is likely to be most rapid, and can help planners compare the cost effectiveness of different actions for enhancing LWD recruitment. As with the preceding example, I mapped reaches with four different levels of recovery potential based on channel slope and bankfull channel width. Recovery potentials for different channels are first classified by slope, with slope >0.04 having low recovery potential and low potential to benefit the salmonid resource. I then classified channels with slope <0.04 into categories of <15 m wide, 15-30 m wide, or >30 m wide, which correspond to approximate recovery times to first recruitment of pool-forming LWD of <35 years, 35-90 years, and >90 years (Table 6-2). These classifications are mapped in the 140 km^2 Finney Creek and 30 km^2 Hansen Creek sub-basins (Figures 6-4 and 6-5).

In addition to mapping recovery potentials, one can also estimate whether channel migration or widening are likely to inhibit the recovery of riparian vegetation and LWD recruitment. As described earlier, channels <15 m wide have high recovery potential due to smaller sizes of pool-forming LWD, as well as a low likelihood of slowed recovery time due to channel migration. Mapping of floodplain reaches for these smaller channels indicates where channel movement could possibly affect recovery of LWD recruitment, but the possibility is remote. Similarly,

Table 6-2. Variation in time to first recruitment of pool-forming LWD and likelihood of slowed recovery time due to channel migration.

Channel width	Time to first recruitment of pool-forming LWD	Likelihood of slowed recovery due to channel migration
< 15 m	<35 years	Low
15-30 m	35-90 years	Low-Moderate
> 30 m	>90 years	High

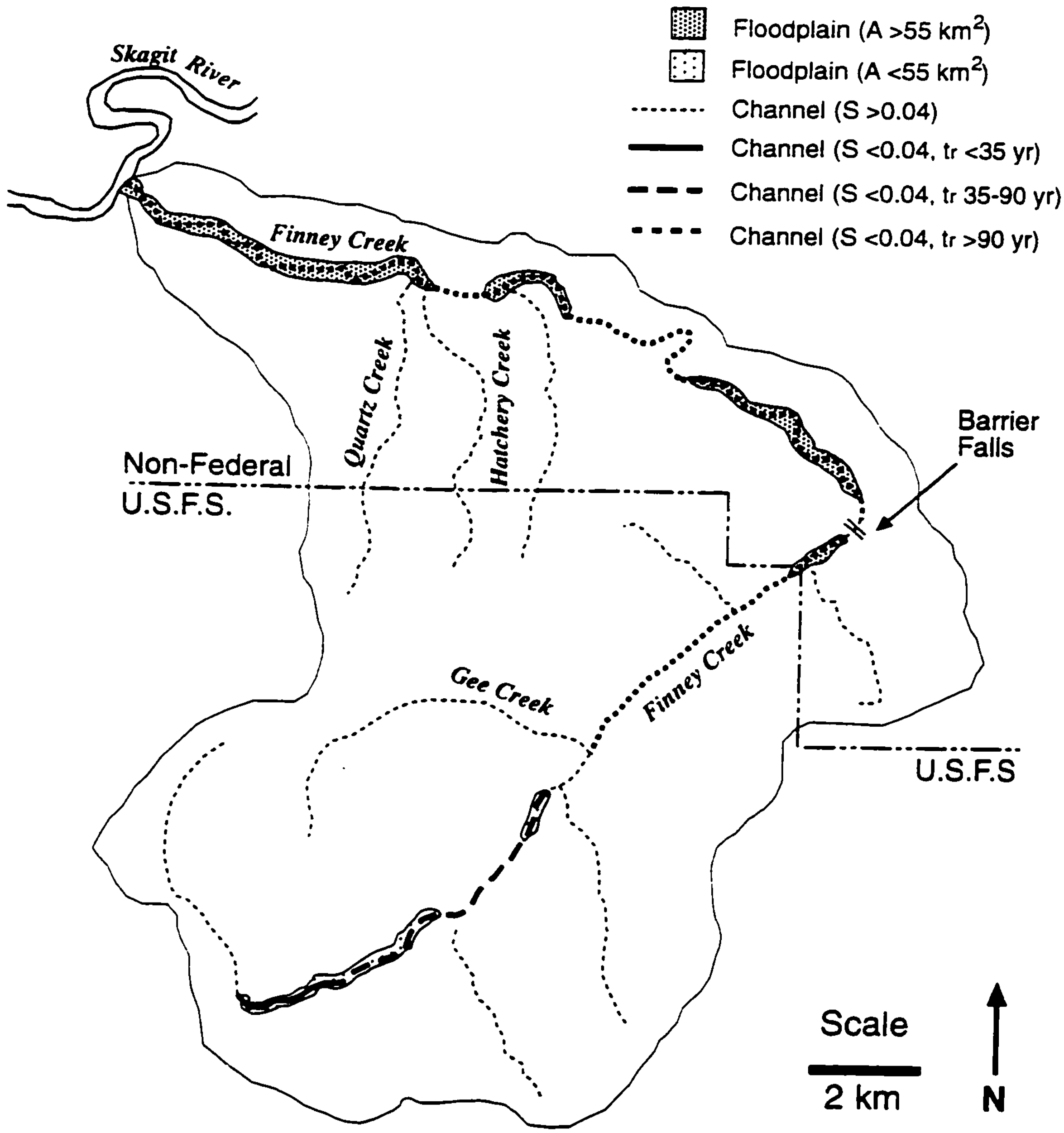


Figure 6-4. LWD recruitment recovery potentials (expressed as number of years from disturbance to first recovery of pool-forming LWD) for the 140 km^2 Finney Creek sub-basin in Skagit County, Washington. Floodplain reaches on channels less than 30 m wide indicate locations where the probability of channel migration affecting recovery of LWD recruitment is low. Floodplain reaches on channels wider than 30 m indicate areas with high probability of channel migration and lengthened recovery time for LWD recruitment. Non-floodplain reaches have extremely low chance of channel migration. (A is drainage area, S is channel slope, and t_r is LWD recovery potential.)

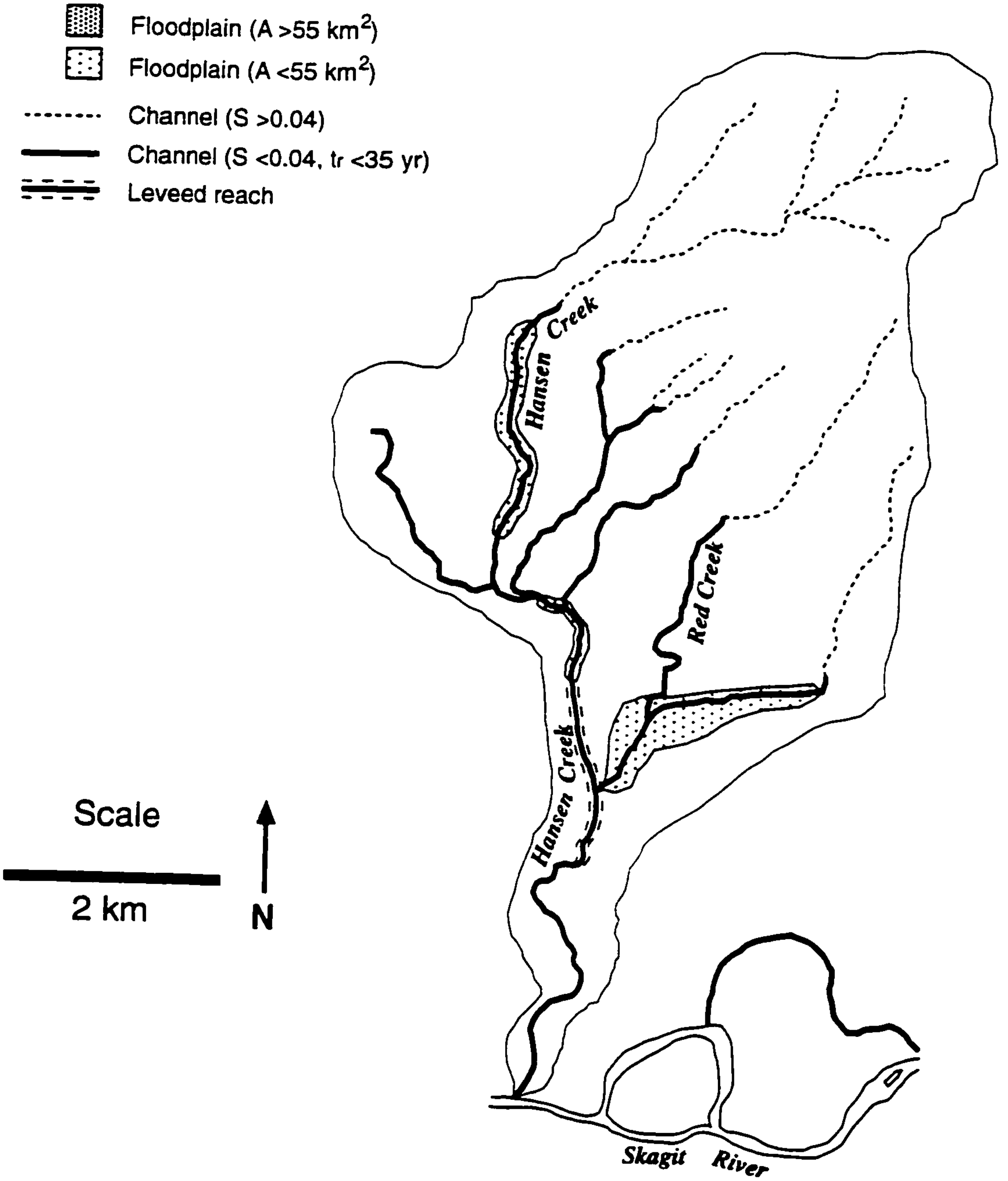


Figure 6-5. LWD recruitment recovery potentials (expressed as number of years from disturbance to first recovery of pool-forming LWD) for the 30 km^2 Hansen Creek sub-basin in Skagit County, Washington. Floodplain reaches on channels less than 30 m wide indicate locations where the probability of channel migration affecting recovery of LWD recruitment is low. Non-floodplain reaches have extremely low chance of channel migration. (A is drainage area. S is channel slope, and t_r is LWD recovery potential.)

mapping floodplain reaches of channels 15 to30 m wide indicates a relatively low probability that channel migration will affect LWD recruitment (Figure 6-4). Larger channels (>30 m wide) have a high probability that channel migration or widening will affect the recovery of riparian forests and LWD recruitment. Where these channels are unconfined (i.e., have floodplains), channel migration is likely and recovery of LWD recruitment is likely to be slower than estimated. Mapping of these floodplains indicates where channel migration will most likely inhibit recovery of riparian vegetation. Large channels confined by valley walls will not have reduced recovery potential due to channel migrations inhibiting riparian forest regeneration (Featherston et al. 1995). Thus, large confined reaches should have recovery potentials that are unaffected by channel movements.

Discussion

In general, reaches of high sediment recovery potential are equivalent to reaches of low LWD recovery potential, and vice versa. That is, small channels tend to have shorter recovery times for LWD recruitment but longer recovery times for sediment supply. Conversely, larger channels have longer recovery times for LWD recruitment and shorter recovery times for sediment supply. In the absence of other considerations, these two factors suggest that restoration of LWD recruitment might focus first on smaller channels in order to generate the greatest change in pool characteristics per dollar spent, whereas restoration of sediment supplies might first focus on larger reaches where recovery will be more rapid.

Nelson (1998) found that effects of altered sediment supply on rearing habitat structure are much more difficult to detect than are effects of altered LWD abundance. This finding suggests that, on the whole, rearing habitats are more strongly affected by changes in LWD recruitment than by changes in sediment supply. Recovery of pool abundance after a decrease in LWD abundance will often take more than 100 years (Table 5-5), whereas recovery of pool depths from increased sediment supply is typically only 10 to 20 years (Table 4-3). Recovery of rearing habitats with respect to these two processes is therefore limited by recovery LWD recruitment. That is, where both processes have been disrupted, pool depths will generally recover within 10 to 20 years, but pool abundance will remain low for several more decades until LWD abundance has recovered. Because decreased pool abundance can result in significant decreases in productive capacity (e.g., Beechie et al. 1994), recovery of fish populations is therefore expected to be limited over the long-term primarily by changes in LWD recruitment.

In the broadest sense, the results of Nelson (1998) and this example suggest that LWD recruitment most limits recovery of rearing habitat productive capacity, and that recovery of sediment supply will recover only a small increment of the total potential productive capacity. Therefore, delays in recovery of LWD recruitment will prolong recovery of rearing habitat conditions. In general, smaller low-slope channels are expected to recover more quickly from decreased LWD recruitment because smaller LWD can form pools, and flooding or increased sediment supply rarely affect riparian forest regeneration in these small channels. Therefore, riparian restoration will be most efficient and most likely to succeed in smaller streams. Larger channels generally require larger LWD to form pools and are more prone to lateral migration during large floods or increased sediment supplies. Thus, larger channels will have slower LWD recovery and a higher likelihood that other factors may inhibit recovery of riparian vegetation. Riparian restoration actions planned for larger channels should therefore consider the likelihood of channel migration before undertaking riparian restoration, and may require additional measures to reduce channel movement and increase the probability that riparian reforestation or management will be successful (e.g., Collins et al. 1994, Abbe et al. 1996).

Riparian restoration actions include a variety of silvicultural actions intended to alter species composition, structure, or growth rates of riparian forests (e.g., Berg 1990, Berg 1995, Berg et al. 1996, Franklin et al. 1996). Some of the most commonly applied actions in riparian forests of the study area may include thinning of stands to increase diameter and height growth, planting of devegetated floodplain areas, or gradual conversion from hardwoods to conifer (Collins et al. 1994). The target stand selected for each silvicultural manipulation should be based on historical reconstruction where possible (e.g., Collins et al. 1994), or at least on sound hypotheses of natural riparian function and stand types where reconstruction is not possible (e.g., Fetherston et al. 1995). The next section provides more detail on assessing silvicultural options for riparian forests.

Where sediment supplies are high or are likely to remain high in the absence of remedial action, riparian restoration may first require reduction of coarse sediment loads. As stated in Chapter 4, most of the change in sediment supply in the study area can be attributed to changes in mass wasting rates. Paulson (1997) found no evidence that soil creep is measurably increased by land use, and also that surface erosion processes account for a relatively small proportion of the coarse bed load supplied to streams in the study area. Therefore, restoration of sediment supply processes focuses on restoring mass wasting rates to approximately natural levels.

Areas prone to mass wasting consist of two general types: roads that have the potential to fail and hillslopes where root strength has been reduced or ground water supplied has been increased. Potential road failure sites commonly include unstable sidecast material, improperly constructed stream crossings, and trapped or misdirected ditch drainage (Collins et al. 1994). Potential hillslope failure sites are typically areas with slope greater than about 35° and reduced root strength (Sidle et al. 1985). Identifying sites prone to failure for both roads and hillslopes requires a systematic method of surveying and classifying sites by their relative hazard levels.

Hazard zonation for mass wasting prone areas has been used extensively in Washington state (Paulson 1997). The product of hazard zonation is a map of landform units that have varying mass wasting types, rates, and sensitivities to land use (WDNR 1995). Map units are derived from a landslide inventory, which indicate areas of concentrated landsliding and help identify landforms that have varying mass wasting rates. The inventory also identifies land uses that most commonly cause increases in mass wasting rates within each landform unit. This information then helps identify modifications to land uses that will reduce mass wasting impacts (e.g., reduced logging or road construction).

Regional limitations of recovery potential estimates

These estimates of recovery potential for sediment supply and LWD recruitment are transferable across regions as long as hydrologic regime and forest growth rates are reasonably similar. Each recovery potential classification is (a) physically based, (b) independent of the magnitude of disturbance, and (c) independent of the natural supply of sediment or LWD. Thus, the basic relationships can be applied to other regions where similar processes occur (i.e., where sediment supply fluctuates and riparian forests provide LWD to the stream). However, each recovery potential is based on a recovery pathway that is free of additional disturbances which may inhibit recovery. Other disturbances that inhibit recovery may be unrelated to the two processes considered here (e.g., large floods which cause bank erosion and inhibit regeneration of riparian forests), or may be interactions between the two (e.g., increased sediment supply causing lateral shifting the channel and inhibiting regeneration of riparian forests).

The basic forms of the recovery potential estimates for both sediment supply and LWD recruitment are transferable across regions because they are based on measurable physical characteristics of streams, and because there are no fundamental regional differences in the physical processes of sediment transport and LWD retention and recruitment. That is, sediment transport rates and annual travel distances can be expressed as some function of stream power, and time to recruitment of functional LWD can be expressed as a function of channel size (which governs the size of pool-forming LWD) and the rate of tree growth in a riparian forest. Of course, each recovery potential estimate can have regionally variable coefficients that are influenced by such factors as dominant precipitation type, bed material composition, or tree species available to colonize riparian forests.

The sediment supply recovery potential is regional in that it is based on data from areas of the western United States where annual hydrographs are dominated by rainfall runoff rather than spring snowmelt. The primary regional difference revealed in Chapter 4 is a difference in the total number of days of sediment transport during a year. Thus, a similar regression form (annual travel distance with respect to stream power) should be applicable anywhere, provided that data for annual travel distance are available to develop a regional equation. Other factors such as the size of bed load material may also influence the coefficients of the equation.

LWD recruitment recovery potential is based on the size of LWD that is stable and forms pools in channels. For any region with similar channel geometry and bed material composition, the relationship developed here should be appropriate. Where channels are typically deeper for a given width, it is likely that larger diameter LWD would be required to initiate pool formation. Conversely, smaller diameter LWD may form pools where channels are shallower for a given width. A relatively small amount of local data can be used to generate a new regional relationship for size of pool-forming LWD as a function of channel size. Pool formation by LWD may also be limited where bed material consists of large particles that are not easily scoured (Andrus et al. 1988), or where there is little sediment stored in the channel (Benda 1994). Furthermore, different tree species have different growth and decay rates, which leads to different estimates of recovery time. The presence of any of these differences should also prompt regional calibrations of the relationships describing the minimum size of stable LWD or time to first recruitment.

Example 2: Enhancing recovery rates of LWD recruitment

The preceding discussion concludes that recovery of LWD recruitment ultimately limits the recovery of physical habitat characteristics in Pacific Northwest streams. Therefore, recovery of LWD recruitment and LWD abundance is a priority for restoration of habitat-forming processes in the study area. This section describes how the LWD recruitment recovery pathways can be manipulated by silvicultural prescriptions, and compares the relative effects of different prescriptions on different channel types. As in Chapter 5, I use Forest Vegetation Simulator (FVS) to model stand growth and Riparian-in-a-Box (RIAB) to model recruitment and function of LWD. These models help estimate how best to enhance recovery of LWD recruitment and where different prescriptions may be most useful or cost-effective. Finally, a summary table summarizes general prescriptions for different channel widths and stand types.

Recovery pathways for LWD recruitment can be described as functions of channel size and stand type, and recovery potential is defined as the number of years from stand initiation to first recruitment of pool-forming LWD (Chapter 5). Recovery time for a stream reach is then the recovery potential minus the current stand age, or the time required for trees to grow from their current size to the size required to influence pool formation in the adjacent channel. Silvicultural manipulations may help reduce this recovery time for LWD recruitment by increasing growth rates of individual trees in the riparian forest to meet a target stem diameter (e.g., Berg 1990, Berg 1995). In this context, riparian restoration would include silvicultural manipulations that are designed to decrease recovery time without reducing the long-term level of LWD recruitment to the stream channel.

The recovery pathways for Douglas fir and red alder from Chapter 5 suggest a general hypothesis for enhancing the recovery time of LWD recruitment. It can be stated: where the mean diameter of trees in the riparian stand is less than the minimum diameter of pool-forming LWD, recovery time can be reduced by thinning operations. Thinning of dense stands typically reduces competition among remaining trees, resulting in increased diameter and height growth. Thus, thinning of stands that are smaller than the target diameter should reduce the time required for trees to reach pool-forming size. However, reduced competition among the remaining trees also reduces mortality, thereby reducing overall LWD recruitment in the near term. This effect would decrease recruitment of pool-forming LWD for several decades if stands were thinned when trees were already large enough to form pools. Together these two effects of thinning imply that silvicultural manipulations must strike a balance between increased growth and decreased mortality.

I first tested this general hypothesis for Douglas fir stands by using the FVS and RIAB models to estimate the effects of various thinning treatments. The first objective of the test was to evaluate whether recovery time was reduced by thinning. That is, did LWD of pool-forming size begin to increase in abundance at an earlier age when the thinning treatment was applied? The second objective was to assess whether cumulative LWD abundance during the first 100 years after treatment was increased by thinning. For this test, initial LWD abundance was set low (0.01 LWD/m) so that the results clearly showed when LWD of pool-forming size began to enter the stream. Douglas fir stands of four different initial quadratic mean diameters were used (initial DBH_q = 12 cm, 23 cm, 38 cm, and 50 cm), and the control (unthinned) and treatment (thinned) scenarios were applied to channels of 5 m, 10 m, 15 m, 20 m, 25 m, and 30m in width. This allowed evaluation of the effect of thinning on stands of differing initial mean diameter by channel size. For each initial diameter and channel size, LWD recruitment after the thinning treatment was compared to the unthinned control, and the result was recorded as negative (thinning produced less LWD than control), positive (thinning produced more LWD than control), or neutral.

The model runs predict that thinning generally does not benefit smaller channels, and causes increased LWD abundance in relatively large channels (Table 6-3 and Figure 6-6). Thinning never increased cumulative LWD abundance in channels less than 15 m wide (e.g., Figure 6-7a). However, thinning tends to increase LWD recruitment when initial DBH_q of the stand is somewhat less than the minimum pool-forming diameter (D_{pf}) of the adjacent channel (Table 6-3 and Figure 6-7b). A plot of diameters at which thinning produced a cumulative LWD abundance similar to that of an unthinned stand shows that thinning is beneficial near larger channels when the DBH_q of the stand is about 10 cm less than the minimum pool-forming diameter for the adjacent channel (Figure 6-6). However, when DBH_q was less than about 25 cm thinning was never beneficial. The most likely explanation for the lack of benefit from thinning along small channels (area "A" on Figure 6-6) is that stocking levels in these scenarios were already below the density at which significant competition occurs. Thus, growth rates are already near maximum, and thinning cannot further improve growth. This phenomenon can be illustrated by plotting the relative density (RD) of stands over time. Relative density is a simple measure of

Table 6-3. Predicted effect of Douglas fir thinning treatment (compared to unthinned control) by initial quadratic mean diameter (DBH_q) and channel width, based on FVS and RIAB modeling of tree growth and LWD recruitment. Plus sign indicates that thinning produced more LWD over the 100-year modeling period than the control, minus sign indicates that thinning produced less LWD than the control, and 0 indicates no difference between treatment and control. Minimum pool-forming diameter (D_{pf}) of LWD is shown in parentheses below each channel width.

	Initial DBH_q			
Channel width	12 cm (5 in)	23 cm (9 in)	38 cm (15 in)	50 cm (20 in)
5 m (D_{pf} = 12.5 cm)	–	–	–	–
10 m (D_{pf} = 25 cm)	–	–	–	–
15 m (D_{pf} = 37.5 cm)	0	0	–	–
20 m (D_{pf} = 50 cm)	+	+	0	–
25 m (D_{pf} = 62.5 cm)	+	+	+	0
30 m (D_{pf} = 75 cm)	+	+	+	+

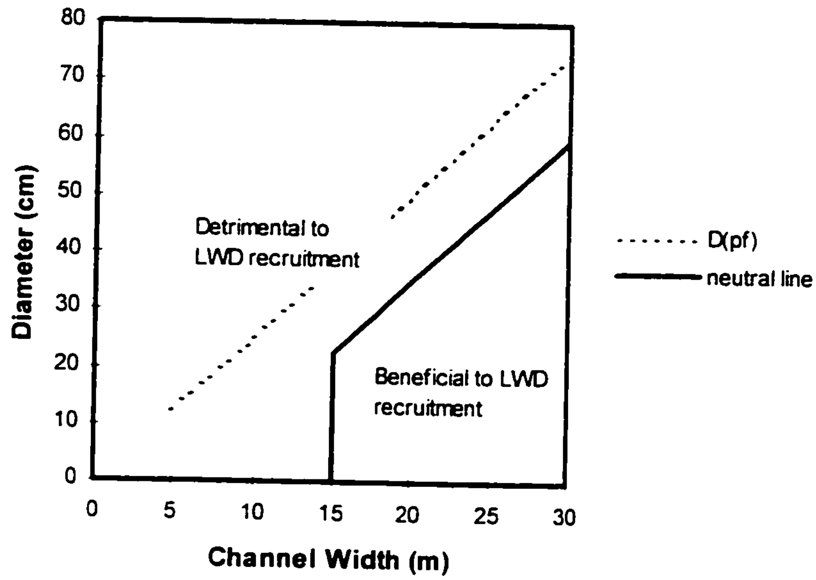
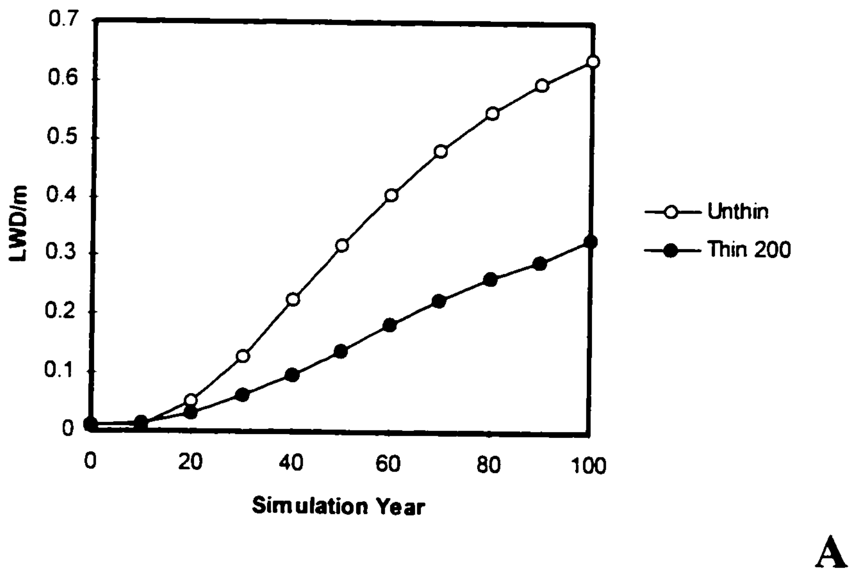


Figure 6-6. Model predictions of quadratic mean diameter (DBH_q) and channel widths where Douglas fir thinning enhances LWD recruitment. Dashed line (D_{pf}) is the minimum pool-forming diameter by channel width. Heavy line (neutral line) represents the predicted DBH_q at which LWD recruitment after a thinning treatment is not significantly different from LWD recruitment for the unthinned case. Combinations of stand diameter and channel width that plot below and to the right of the neutral line represent cases where thinning should enhance recovery of LWD recruitment. Combinations of stand diameter and channel width that plot above and to the left of the neutral line represent cases where thinning will inhibit recovery of LWD recruitment.



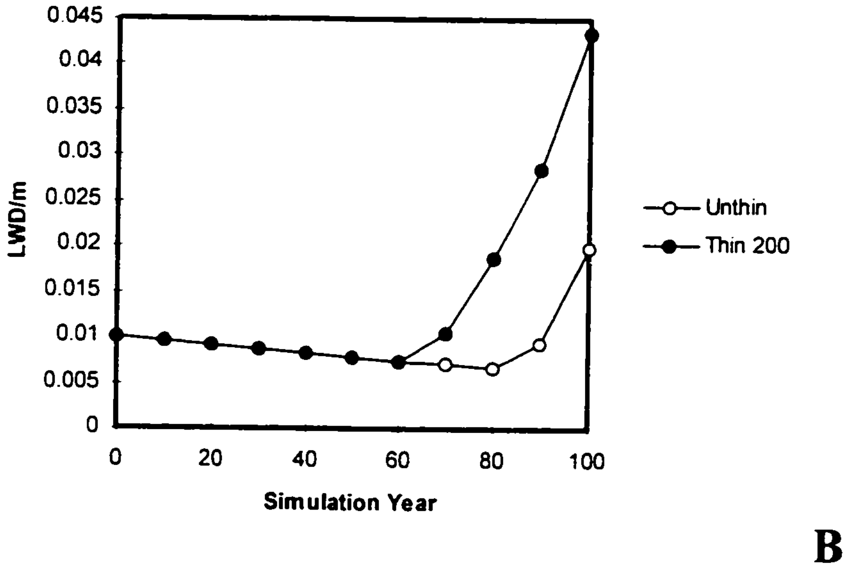


Figure 6-7. Examples of predicted LWD abundance over time for model runs with 23 cm (9 inch) DBH Douglas fir for (A) a channel 10 m wide, and for (B) a channel 20 m wide. The "Unthin" stand has no thinning treatment: the "Thin 200" stand was thinned to a 440 trees per hectare (200 trees per acre). Note different y-axis scales.

relative stand density that ranges from zero to a species-dependent maximum value (Curtis 1982), which is correlated with the level of competition among trees in the stand. For Douglas fir, RD is calculated as

$$RD = \frac{G}{\sqrt{DBH_q}}.$$

where G is basal area. When RD is less than about 7 (~50 in English units), competition is insignificant and the rate of diameter growth is high while mortality rate is low. An RD of 9.5 (~65 in English units) corresponds to a "normal" stand density with some competition among individual trees, and 14 (~100 in English units) appears to be the approximate biological maximum for Douglas fir (Curtis 1982).

The initial stocking level for the Douglas fir seedling scenarios modeled here was 1075 trees per ha (435 trees per acre), which is a typical planting density for tree farms in the study area. At these stocking levels, competition between individual trees is insignificant until approximately age 30, by which time the projected DBH_q is 22 cm and RD is about 9 (Figure 6-8). Thus, when thinning was modeled for a 12 cm DBH_q stand, the projected RD of the stand was only 4, competition among trees was already low, and thinning did not increase diameter growth. Thinning of small diameter stands did not increase the abundance of LWD delivered to any channel until channel width exceeded 15 m, primarily because there was no significant decrease in time to first recruitment until the minimum pool-forming diameter exceeded 38 cm. For a channel 15 m wide, RD of the unthinned stand reached 9 when the stand had a DBH_q of 22 cm, and growth slowed for the next few decades until DBH_q reached 38 cm (Figure 6-8). Thinning of the young stand delayed competition by about 10 years (i.e., the RD of 9 was not reached until DBH_q was 31 cm), and reduced the time to first recovery of pool-forming LWD by just under 10 years.

Thinning of Douglas fir to enhance recovery of LWD recruitment should typically occur when DBH_q is less than the minimum pool-forming diameter, as long as RD is greater than about 9 at the time of thinning. Thinning at this point in the development of the stand can reduce the stem density to the point that RD remains below 9 for three to four decades. Reduced RD decreases competition among the remaining trees, causing increased diameter growth and decreased mortality until DBH_q is sufficient to provide pool-forming LWD. By the time DBH_q is approximately equal to the minimum diameter of pool-forming LWD, RD has increased again to

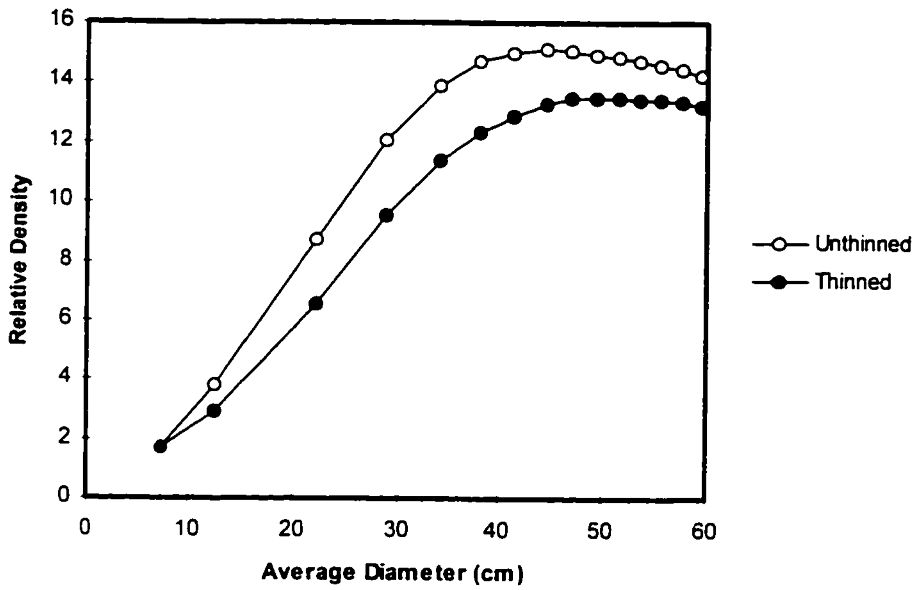


Figure 6-8. Simulated change in relative density as average diameter increases over time. For a channel 15 m wide, RD of the unthinned stand reached 9 when the stand had a DBH_q of 22 cm, indicating that significant competition within the stand is occurring by that time. Thinning of the young stand delayed significant competition (i.e., RD remained less than 9) until DBH_q was 31 cm, which reduced the time to first recovery of pool-forming LWD by just under 10 years.

about 8, which increases mortality and the projected recruitment of LWD. As guide for enhancing recovery of LWD recruitment in Douglas fir stands, Figure 6-6 illustrates that thinning increases cumulative LWD abundance during the first 100 years when DBH_q is below the "neutral" line, and decreases cumulative LWD abundance when DBH_q is above the neutral line.

The location of the neutral line is most dependent on the "target" LWD diameter for a given channel width. The scenarios shown here use the approximate minimum pool-forming diameter as the target (dashed line in Figure 6-6), which gives the best model approximation of total LWD abundance by channel width, and most accurately compensates for the uniform depletion rate across the ranges of LWD and channel size (see Appendix B). Use of other diameter targets will result in a shift in the neutral line location, and will suggest different management prescriptions. For example, changing to a larger target diameter such as the average diameter of LWD in old-growth streams (e.g., Bilby and Ward 1989) will cause the neutral line to shift upward and to the left. This shift would put more scenarios into the "beneficial to LWD recruitment" category, and allow for more riparian management. However, the models used here predict that all of the additional scenarios would result in delayed recovery of LWD and pool abundance.

One might also expect the neutral line location to be dependent on site class, which is a measure of the growing potential of a site. As noted in Appendix B, most riparian forest sites in the study area are classified as Site II for Douglas fir (height of 170 feet at age 100, McArdle et al. 1961), and relatively few sites are classed as Site I (200 feet) or Site III (140 feet). I surmised that the neutral line location for Site I scenarios would shift downward and to the right because higher growth rates would reduce the benefit of thinning to attain larger trees. Conversely, I expected the neutral line to shift upward and to the left for Site III scenarios because slower growth rates would increase need for thinning to attain larger trees in less time. However, based on Douglas fir scenarios for Site Classes I, II and III (Jeff Welty, Weyerhaeuser Company, unpublished data from DFSIM modeling), changes in location of the neutral line due to site class were negligible because thinning regimes only alter growth rates within the limits of site potential. Thus, changing site class did not alter the location of neutral line in Figure 6-6, but site class had a large effect on the basic recovery rate for LWD recruitment. First recruitment of pool-forming LWD occurred about a decade later on Site II than on Site I, and about two decades later on Site III than on Site I.

The general hypothesis that thinning may decrease recovery time for LWD recruitment may also apply to red alder stands. However, recruitment of pool-forming LWD will typically occur sooner from red alder stands than from Douglas fir stands because diameter and height growth are much faster during the first several decades (Figure 5-6). Thus, there are fewer opportunities for thinning to benefit LWD recruitment. Moreover, red alder forests are short-lived compared to conifer forests (less than 130 years compared to several hundred years), and shrub vegetation often follows the death of alder. Because tree species are often absent after the death of alder, LWD recruitment and many other riparian functions (such as shade) may be essentially zero over the long-term. Therefore, managers interested in long-term riparian function should consider conversion to conifer species in those areas where conifer tended to dominate under natural conditions. For all practical purposes, this includes channels less than about 30 m wide, and may include channels up to 40 or 50 m wide (Collins et al. 1994, Abbe and Montgomery 1996). More specific location of such areas requires searching historical records such as Gannet (1899) and Ayers (1899), as well as other available maps and notes.

Where conversion from red alder to conifer is desirable, managers may consider different options ranging from no action to clearing the alder stand and replanting with confer species. One intermediate option is thinning the alder to about 440 trees per hectare (200 trees per acre) and planting conifer. The following model runs compare these three options for different channel widths in order to identify the management actions most likely to benefit LWD recovery. Thin and plant scenarios are modeled with shade tolerant western red cedar and western hemlock seedlings as the planting stock. Clear and replant scenarios are modeled with Douglas fir seedlings as the planting stock.

Thinning of red alder does not benefit LWD recruitment during the first 100 years for any mean stand diameter when channel width is less than 20 m (Table 6-4). However, thinning generally benefits LWD recruitment when quadratic mean diameter of the stand is less than 25 cm and channel width is $\geq$20 m. for channel widths greater than 20 m, thinning appears to increase total LWD recruitment during the first 100 years after treatment as long as the quadratic mean diameter of the stand is at least 25 cm (10 in) less than the minimum pool forming diameter (Figure 6-9).

The model runs indicate that thinning of red alder is more often detrimental to LWD recruitment during a 100-year modeling period than is thinning of Douglas fir (compare Figures

Table 6-4. Predicted effect of red alder treatments (compared to unthinned control) by initial quadratic mean diameter (DBH_q) and channel width, based on FVS and RIAB modeling of tree growth and LWD recruitment. Plus sign indicates that thinning or clearing and replanting with Douglas fir produced more LWD over the 100-year modeling period than the control, minus sign indicates that a treatment produced less LWD than the control, and 0 indicates no difference between treatment and control. Minimum pool-forming diameter (D_{pf}) of LWD is shown in parentheses below each channel width. Where "thin" appears in parentheses below a plus sign, the thinning treatment benefited LWD recruitment during the 100-year simulation whereas the clear and replant treatment did not.

Channel width	Initial DBH_q 5 cm (2 in)	 25 cm (10 in)	 48 cm (19 in)
5 m (D_{pf} = 12.5 cm)	-	-	-
10 m (D_{pf} = 25 cm)	-	-	-
15 m (D_{pf} = 37.5 cm)	-	-	-
20 m (D_{pf} = 50 cm)	+ (thin)	+ (thin)	-
25 m (D_{pf} = 62.5 cm)	+ (thin)	+ (thin)	-
30 m (D_{pf} = 75 cm)	+ (thin)	+ (thin)	0

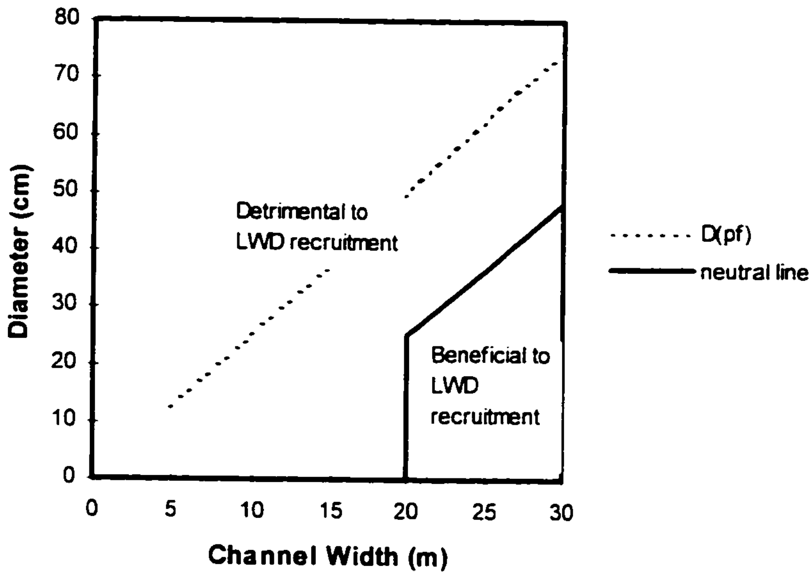


Figure 6-9. Model predictions of quadratic mean diameter (DBH_q) and channel widths where red alder thinning enhances LWD recruitment during the first 100 years after treatment. Dashed line (D_{pf}) is the minimum pool-forming diameter by channel width. Heavy line (neutral line) represents the predicted DBH_q at which LWD recruitment after a thinning treatment is not significantly different from LWD recruitment for the unthinned case. Combinations of stand diameter and channel width that plot below and to the right of the neutral line represent cases where thinning should enhance recovery of LWD recruitment. Combinations of stand diameter and channel width that plot above and to the left of the neutral line represent cases where thinning will inhibit recovery of LWD recruitment. Note that this figure ignores recruitment more than 100 years after treatment, and that all scenarios converting alder to conifer will have greater LWD recruitment over longer time periods because red alder recruitment virtually ceases after about age 100.

6-6 and 6-9). The main reason for this is that red alder has a higher growth rate than Douglas fir. and thinning is less likely to reduce the time required for trees to reach pool-forming diameter. However, these model runs do not account for differences in LWD recruitment more than 100 years after a treatment. Therefore, they do not adequately illustrate longer term (i.e., >100 years) patterns of LWD recruitment. Because alder recruitment typically ceases after about 100 to 130 years, total LWD recruitment will decline to near zero as the stand is replaced by shrub species. In the absence of a treatment that converts the stand to conifer, LWD recruitment during the second century of an untreated alder stand will be extremely low. Thus, either conversion treatment (thinning and planting conifer, or clearing and planting conifer) should result in greater LWD recruitment during the second century.

Where conversion to conifer is desired, thinning and conifer planting in stands below and to the right of the neutral line in Figure 6-9 should show increased LWD recruitment in both the near-term and long-term because thinning is beneficial to recruitment of pool-forming alder LWD in the first century, and conversion to conifer provides a long-term source of LWD. For stands that are above and to the left of the neutral line, thinning and planting with conifer will inhibit near-term alder recruitment, but should still benefit longer term conifer recruitment. In such cases, managers must attempt to weigh the benefit of improved long-term LWD recruitment (more than 100 years from now) against near-term losses in LWD recruitment.

Identifying potential riparian restoration actions

Figures 6-6 and 6-9 can be used to help identify sites where active riparian management is likely to enhance the recovery LWD recruitment. Based on these two figures, we can identify four generalized riparian scenarios and evaluate management options for each (Table 6-5). The four scenarios are:

(1) Douglas fir stands where thinning inhibits near-term recovery of LWD recruitment,

(2) Douglas fir stands where thinning improves near-term recovery LWD recruitment,

(3) red alder stands where thinning and conversion to conifer inhibit near-term recruitment but enhance long-term recruitment, and

(4) red alder stands where thinning and conversion to conifer improve both near-term and long-term LWD recruitment.

Table 6-5. Description of generalized riparian management scenarios for Douglas fir and red alder based on Figures 6-6 and 6-9. Channel width and stand type scenarios identify combinations of channel width, tree species, and average diameter of stand (DBH_q). Effect of management indicates how a management option effects near-term (first 100 years) and long-term (beyond 100 years) LWD recruitment. Preferred option indicates the best management strategy for LWD recruitment based on model results.

Channel width and stand type	**Effect of management**	**Preferred option for LWD recruitment**
Douglas fir stands		
Channel width <15 m, All Douglas fir stands *or* Channel width >15 m, Douglas fir with DBH_q exceeding D_{pf}–10, where D_{pf} = minimum pool-forming diameter of LWD (in centimeters)	Thinning inhibits near-term LWD recruitment, but does not effect long-term LWD recruitment	No action
Channel width >15 m, Douglas fir with DBH_q less than D_{pf}–10, where D_{pf} = minimum pool-forming diameter of LWD (in centimeters)	Thinning improves near-term LWD recruitment, and does not inhibit long term recruitment	Thin to increase diameter and height growth
Red alder stands		
Channel width <20 m, All red alder stands *or* Channel width >20 m, red alder with DBH_q exceeding D_{pf}–25, where D_{pf} = minimum pool-forming diameter of LWD (in centimeters)	Thinning and planting conifer inhibits near-term red alder recruitment, but stand is eventually replaced by conifer species leading to increased long-term LWD recruitment No action yields higher near-term red alder recruitment, but stand is eventually replaced by shrub species leading to reduced long-term LWD recruitment	To improve long-term recruitment at the expense of near-term recruitment: thin alder and plant red cedar and western hemlock For higher near-term recruitment at the expense of long-term recruitment: no management
Channel width >20 m, red alder with DBH_q less than D_{pf}–25, where D_{pf} = minimum pool-forming diameter of LWD (in centimeters)	Thinning and planting conifer enhances both near-term and long-term LWD recruitment	Thin alder and plant red cedar and western hemlock

Among these four combinations of channel width and stand type, three (numbers 1, 2, and 4 in the previous list) have a single preferred management option based on the model results. All Douglas fir stands along channels <15 m wide should remain unthinned because thinning typically reduces near-term (first 100 years) LWD recruitment. For larger channels ($\geq$15 m wide), Douglas fir stands with DBH_q significantly less than the minimum pool-forming diameter for the adjacent channel should be thinned to increase diameter and height growth, which is projected to increase LWD recruitment in the near-term and should not inhibit LWD recruitment over the long term (beyond 100 years). Red alder stands along channels >20 m wide can be thinned and converted to conifer (modeled as thinning and interplanting 110 trees per hectare each of western red cedar and western hemlock), which should increase near-term alder recruitment as well as long-term conifer recruitment.

The fourth scenario presents a problem for managers because no single option is best for both near-term and long-term LWD recruitment. In this case, managers must weigh the benefits of improving long-term conifer LWD recruitment against a decrease in near-term alder recruitment. Where this scenario is widespread in a channel network, a prudent approach may be to thin and convert some of the stands in order to provide sources of long-term conifer LWD recruitment, while maintaining other stands as sources of near-term alder LWD recruitment. In areas where there are few stands in this category, thinning and conversion of all stands of this type may be appropriate if other stand types can sustain near-term LWD recruitment.

In order to efficiently plan riparian restoration actions, current riparian stand types can be first identified by aerial photograph interpretation. Stand types should be delineated based on average DBH (indicated by the size of tree crowns on aerial photos) and species composition (indicated by crown shape on aerial photos). When possible, the target stand type for riparian forest management should be based on stand types in natural riparian zones, which requires historical reconstruction (e.g., Collins et al. 1994). Under natural conditions, riparian forests in the study area are usually one of four types: silver fir zone forests or western hemlock zone forests where disturbance is not recent (Franklin and Dyrness 1973), or red alder or black cottonwood stands in recently disturbed areas. Silver fir zone forests typically include western hemlock and silver fir, and western hemlock zone forests typically include Douglas fir, western hemlock, and western red cedar. Prior to wide-spread settlement in the Skagit River basin, hardwood and conifer mixed forests predominated on the lower mainstem river, and conifer forests predominated elsewhere (Gannett 1899).

Figure 6-10 illustrates the identification of riparian management options in lower Illabot Creek based on stream size (average bankfull width in 1996, from Beamer et al. 1998) and forest type. The drainage area of Illabot Creek is 115 km^2, and all channels in the lower reaches have an average bankfull width greater than 20 m except the alluvial fan reach which has several smaller channels. The riparian corridor (here defined as an area 60 m on either side of the channel) consists of a mixture of pasture and second growth forest. Forest types are classified as in Collins et al. (1994), where species mixes are classified as deciduous (>70% deciduous), conifer (>70% conifer), or mixed, and seral stages are open (grasses or shrubs), brush (<15 cm DBH), sapling (15 cm to 30 cm DBH), or pole (>30 cm DBH). Historically, lower Illabot Creek was most likely dominated by conifer riparian forests (based on Gannett 1899).

Most reaches >20 m wide are bordered by sapling deciduous or mixed stands, which can be thinned and converted to conifer in order to improve near-term and long-term LWD recruitment. The two objectives of the treatment are (1) improve growth of the alder which should increase LWD recruitment during the first 100 years (Figure 6-9), and (2) release growing space for conifer which should increase LWD recruitment after the first 100 years. The alluvial fan reaches are bordered by similar stands (sapling mixed), but do not necessarily warrant similar management. These channels are less than 20 m wide, and thinning of these stands is likely to inhibit LWD recruitment during the first 100 years. Moreover, channel migration and avulsion are common on the alluvial fan landform, which means that treatments to establish conifer have a relatively high risk of failure. The young conifer stands on the north side of the stream are smaller than 15 cm diameter and border channels that are >15 m wide, which suggests that thinning will improve LWD recruitment (Figure 6-6). Open areas on the south side of the channel should be planted to conifer in order to initiate recovery of LWD recruitment.

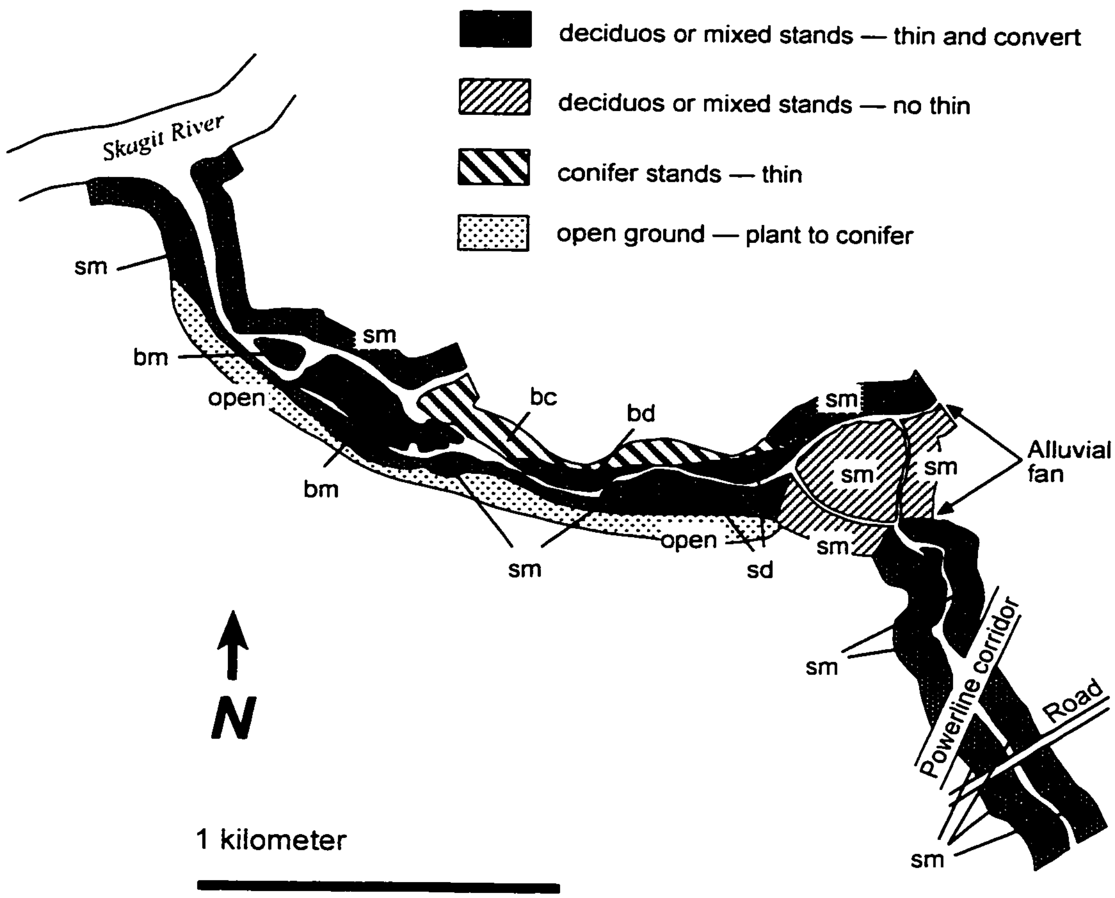


Figure 6-10. Riparian restoration options for the lower reaches of Illabot Creek (drainage area 115 km^2). See text for specific descriptions of restoration actions.

Chapter 7. Summary and Conclusions

Beechie et al. (1994) noted that the development of a realistic restoration plan requires an understanding of (1) expected recovery endpoints based on historical reconstruction, (2) causes of habitat degradation, (3) the time required for the resource to recover, and (4) the degree to which restoration efforts can speed recovery (Sedell and Luchessa 1982, Allen and Hoekstra 1987, Poff and Ward 1990, Reeves et al. 1991, Cairns 1990). The first two elements are essential to identifying the array of actions needed to restore habitat-forming processes in a watershed. Historical reconstruction provides a realistic indication of the natural potential of different parts of the landscape and channel network, and helps focus restoration efforts by eliminating options that attempt to create conditions that are outside the "potential" of restoration site. Historical reconstruction also helps understand the causes of degradation, which typically require at least minimal assessments of processes that affect habitat conditions in stream channels (e.g., riparian functions, sediment supply processes). The second two elements give resource managers realistic expectations for restoration actions, and help focus on the most cost-effective restoration options available.

A main assumption in this dissertation is that recovery of biological populations or communities is relatively rapid compared to the recovery of the physical habitats that have been altered by disrupted sediment supply or LWD recruitment. This is supported by the literature review of Niemi et al. (1990) which found that fish populations generally recover within about three years once the habitat disturbance has been removed. Where recovery of populations was much longer, habitats had been permanently altered (e.g., channelized) or watershed processes had been disrupted (e.g., from logging) (Niemi et al. 1990, Yount and Niemi 1990). This suggests that fish populations cannot recover until habitat conditions have been restored, and that limiting habitats are a logical focus of habitat restoration efforts. Further support for this conclusion is found in the Skagit River study area where salmonid production has significantly recovered within the first year after isolated habitats were reopened to migrating salmon (Beamer et al. 1998). In this case, fish production from isolated habitats in a Baker Lake tributary was limited for many years by lack of access. Once the habitat constraint was removed, salmonid utilization significantly increased during the first year. This suggests that the recovery time for salmonids is negligible compared to that of habitats that require several decades to recover, and that recovery of salmonid production is in part limited by the recovery of physical habitat conditions.

Given that a focus on habitat is appropriate, Chapter 2 described a strategy for diagnosing and treating anthropogenic disturbances to habitat-forming processes. The strategy is driven primarily by the recognition that (1) salmonids are adapted to local environmental conditions, and (2) spatial and temporal variations in landscape processes create a dynamic mosaic of habitats in a river network. The former implies that recovery endpoints could logically be based on natural conditions occurring in channel network. However, the latter suggests that identifying "natural" habitat conditions is difficult because there is a wide range of possible states in the natural environment and habitat conditions naturally change over time. The strategy avoids these difficulties by adopting a goal of restoring and maintaining habitat-forming processes (Beechie et al. 1996), which directly addresses causes of degradation rather than focusing on conditions in stream channels. This goal is consistent with ecosystem management concepts (i.e., does not focus attention on individual species), and avoids focusing restoration efforts on the creation of specific habitat conditions that may not be appropriate for many locations in a channel network. Therefore, it is expected to guide restoration efforts toward actions that are more effective and sustainable.

Under the goal, restoration actions are identified by assessing changes in habitat-forming processes due to land use. Where processes have been significantly altered from their natural rates (based on historical reconstruction or other estimates of natural rates), restoration actions can be identified and planned. Once restoration actions are listed, the strategy allows for prioritization of those actions. Prioritization can be based on local management objectives (e.g., endangered species, aquatic community structure, etc.), and may often use cost-effectiveness assessments as a guide in ranking restoration actions. However, it is important to remember that prioritization is always subordinate to the goal of restoring habitat-forming processes, and that prioritization does not alter the types of restoration actions that may be taken. Rather, it alters the sequence in which restoration actions are completed.

Enactment of the strategy requires an understanding of natural rates of habitat-forming processes, the degree to which those processes have been altered, and rates and pathways of recovery for those processes. Toward this end, this dissertation focuses on the description of disturbance mechanisms and recovery pathways for two important habitat-forming processes in western Washington: sediment supply and LWD recruitment. These processes were selected for study primarily because there is good evidence that rearing habitats are a key factor limiting production of several salmonid species in and near the Skagit River basin, and that significant losses in rearing habitat have occurred in the past century (e.g., Beechie et al. 1994, Collins et al. 1994). Thus, a focus on restoring physical habitat conditions in Pacific Northwest streams is a logical component of restoration planning.

For the study area, the isolation of slough and tributary habitats account for the largest known losses in coho salmon smolt production capacity in the Skagit River basin (Beechie et al. 1994). These problems required little additional research before restoration could begin, and Skagit System Cooperative began cooperative efforts with other state and federal agencies to reconnect isolated habitats. Remaining habitat losses result primarily from changes in riparian functions and sediment supply, which are well understood causes of habitat degradation in the Pacific Northwest (e.g., Sidle et al. 1985, Bilby and Ward 1991, Collins et al. 1994). However, recovery pathways for these processes are less well studied, and cost-effective restoration actions cannot be accurately identified without an understanding of the rates and pathways of recovery (see Cairns 1990). Hence, this dissertation has attempted to advance our understanding of the rates and pathways of habitat recovery (and therefore of salmonid recovery) where physical habitats have been altered by changes in sediment supply and LWD recruitment.

Chapter 4 focused on describing the mechanisms driving disturbance and recovery of sediment supply, whereas Chapter 5 focused on disturbance and recovery of LWD recruitment. The primary objectives of each chapter were to describe (1) the magnitude and duration of effects of land use on each habitat-forming process, and (2) the magnitude and duration of effects of each altered process on stream channel morphology. Together they describe the linkages between a land use and its effect on channel morphology, which helps to identify the causes of degraded conditions and to devise restoration actions having a greater likelihood of success. Each process was first described at the site scale in order to best explain the disturbance mechanisms and the rates and pathways of recovery. I then discussed landscape-level patterns of disturbance and recovery for each process, which provided a means of assessing the cumulative impact of multiple site-level disturbances on a channel network.

Sediment supply in the study area is most affected by changes in mass wasting rates, which were typically moderated by dense forests in the natural environment. Forest fires infrequently killed large forest patches, leading to decreased rooting strength and increased mass wasting rates for two or more decades. Averaged over time and space, natural fires in the study area should have maintained 5% to 10% of the watershed in stands less than 20 years old. These young stands appear to have an average mass wasting rate approximately four times that of mature forest areas (Paulson 1997). Under typical forest management regimes on state and private lands, the percentage of forest area <20 years old would increase to an average of about 33% for an average rotation length of 60 years. Additionally, roaded areas have mass wasting rates approximately 40 times that of mature forest areas (Paulson 1997), primarily because fill slopes and stream crossing fills are prone to failure on roads constructed to logging standards of the past several decades. Roaded areas occupy an average of 2% of the watershed under present forest management regimes. The combined effect of these land uses tends to increase average annual sediment supply to more than twice that of the natural fire regime (Table 4-1).

Average annual sediment supply is useful as an indicator of the landscape-level disturbance to sediment supply in a watershed, but it does not accurately reflect the episodic nature of mass wasting inputs to stream channels. Mass wasting rapidly delivers large quantities of sediment to stream channels, which typically causes rapid aggradation and/or widening of a channel. Effects on habitat conditions include shallowing or elimination of pools, and possible reductions of riparian functions such as shading or LWD recruitment. Export of sediment from an aggraded reach by fluvial transport is more gradual than the inputs, and recovery of habitat conditions is also gradual. The recovery pathway is the fluvial transport of sediment through and out of an aggraded reach, and the rate of recovery can be described in terms of sediment transport rates. I selected the average annual travel distance of sediment as the most appropriate indicator of sediment transport rate, and described the ability of a stream to recover (or recovery potential) in terms of that rate. I estimated the average annual travel distance for different streams based on an empirical relationship between stream power and average annual travel distance, as well as on a simpler relationship between channel width and average annual travel distance for channels with slope less than 0.03. The empirical data indicate that average annual travel distance averages about 20 times the channel width for low slope channels in the study area.

Mapping of recovery potential for sediment supply gives an indication of where stream channels (and habitat conditions) are likely to recover from disturbances most rapidly. Such maps can help in restoration planning by indicating where restoration of sediment supply will tend to be most cost-effective. Where sediment budgets indicate that sediment supplies are elevated (e.g., Paulson 1997), restoration funds might best be focused in watersheds with greater extent of streams with high recovery potential in order to achieve the greatest return on investment. A limitation of recovery potential is that it does not directly indicate the actual recovery time, which is dependent on both the recovery potential (average annual travel distance) and the degree of disturbance (quantity of sediment to be evacuated). For sediment supply, the amount of increased sediment in a channel is difficult to quantify, and actual recovery time can therefore be estimated only in rare cases (e.g., Lisle 1982, Collins et al. 1994).

LWD recruitment has most often been altered by previous logging practices, although channel widening has also been a cause of riparian disturbance in some cases. Where riparian forests have been removed, LWD abundance in the stream channel gradually decreases due to depletion (i.e., decay and export), and abundance continues to decrease until the riparian forest has recovered to the point that LWD recruitment rate exceeds the depletion rate. For this dissertation, the process of LWD recruitment was considered disturbed whenever the size of LWD available for recruitment was smaller than the minimum size of LWD required to form a pool. On average, forest management (assuming an average rotation of 60 years) was projected to reduce recruitment of LWD large enough to form pools by about 35% to 100% compared to the natural fire regimes in the study area (Tables 5-1 and 5-2).

Effects of altered LWD abundance on pool formation are well documented in the literature (e.g., Bilby and Ward 1989, Montgomery et al. 1995, Beechie and Sibley 1997). In relatively small streams (bankfull width less than 15 or 20 m), reduced LWD abundance typically results in fewer pools and less total pool area. In larger streams, pool area is less significantly related to LWD abundance, but LWD abundance is related to number of pools. LWD-formed pools also tend to be deeper than free-formed pools on average (Abbe and Montgomery 1996), so decreased LWD abundance may also result in shallower average pool depths in a reach. These findings suggest that decreased LWD abundance resulting from removal of riparian forests will also result in decreased pool abundance and depth, and also that recovery of LWD recruitment and abundance will be accompanied by increased pool abundance and depth.

Recovery pathways for LWD recruitment are primarily a function of the tree species colonizing (or planted in) a disturbed riparian area. This dissertation focused on describing the recovery pathways for two common riparian species in the Skagit basin study area: red alder and Douglas fir. Each has different rates and patterns of growth, which affect the size and amount of LWD recruitment at any point in time during riparian forest recovery. In general, red alder grows most rapidly during the first several decades after disturbance, but most trees have died by age 100. Douglas fir grows more slowly, but exceeds alder in size after about 60 years and persists well beyond 100 years.

Rates of recovery for LWD recruitment depend on the riparian recovery pathway and the size of the adjacent stream. Along small streams (in which smaller LWD can form pools), recovery of pool-forming LWD is more rapid for alder stands, but LWD recruitment declines rapidly after 100 years as the stand dies. Recovery is slower for Douglas fir stands, but LWD recruitment can be sustained for centuries. Along larger channels, red alder does not attain sufficient size to form pools, so recovery of recruitment of pool-forming LWD does not occur. By contrast, Douglas fir stands exceed pool-forming size after several decades, and pool-forming LWD can be sustained for centuries.

Either of these recovery pathways can be expressed in terms of time to first recruitment of pool-forming LWD (recovery potential), which is a function of stand type and channel size. Hence, recovery potentials for either pathway can be mapped, giving an indication of where riparian recovery may be most rapid. As smaller channels have the shortest average recovery time, they are considered to have the greatest recovery potential. However, this was not a reliable indication that active management of riparian forests would most effectively increase recovery rates in small streams. In order to estimate where active management of riparian forests may be most beneficial in terms of LWD recruitment, I modeled stand development and LWD recruitment for different silvicultural prescriptions applied to the two recovery pathways. In general, the models predicted that active management of riparian forests is unlikely to improve LWD recruitment on any channel less than 15 or 20 m wide. In these channels relatively small LWD can form pools, the trees reach pool-forming size rapidly, and thinning simply tends to reduce the rate of recruitment of adequately sized wood. LWD recruitment to larger channels is more likely to benefit from thinning of riparian forests because thinning reduces competition among trees and increases growth rates. Thus, trees attain pool-forming size more rapidly and LWD recruitment recovers more quickly.

Nelson (1997) showed that changes in LWD abundance typically have a greater effect on pool-formation and rearing habitat conditions than do changes in sediment supply. Furthermore, several studies indicate reduced levels of LWD abundance (compared to average natural levels) has a more widespread impact on stream channels than does increased sediment supply (e.g., WDNR 1994). Thus there are at least preliminary indications that recovery of LWD abundance is likely to limit overall recovery of physical habitat conditions in the study area. At first glance, this implies that restoration efforts should focus on riparian forests prior to addressing sediment supply. However, increased sediment supply can inhibit riparian recovery, and may also increase the risk of failure of riparian restoration efforts.

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Appendix A. Wildfire regimes in the western hemlock and Pacific silver fir zones

Gannett (1899) and Ayers (1899) described forest conditions of Washington based on numerous timber cruises, maps and anecdotal descriptions recorded prior to the turn of the century. Gannett (1899) described the western portion of Skagit county, which covers at least 4 of the study basins, as "entirely forested except for a considerable tract of prairie about the mouth of the Skagit River". He further stated that there had been no "fires of magnitude" in the county, and that logging had cleared virtually all lands near the coast and in the immediate valley of the Skagit River up to the Sauk River. Ayres (1899) described the Forest Reserve of Washington (which covers the eastern part of the study area and the remaining 6 study basins) as being dominated by Douglas fir generally to an elevation of 2000 feet, and then silver fir and western hemlock from 2000 to 4000 feet. The valley bottoms (apparently referring to the flood plain and low terraces of the Skagit and Sauk Rivers) were described as having large trees of Sitka spruce, cedar, big-leaf maple and cottonwood, with a dense tangle of undergrowth that included alder, vine maple, devil's club and ferns. In the red (Douglas) fir zone, western red cedar and Sitka spruce were also common but usually subordinate to Douglas fir. In some wetter areas and along streams in the red fir zone, spruce or cedar was the dominant species. In the drier region of the Upper Skagit River and the Granite-Ruby Creek basin, scrub pine were common.

Floodplains of large rivers tend to have patchy stands of varying ages, which are caused primarily by river channel migration and avulsion combined with the effects of large stable log jams (Agee 1988, Abbe and Montgomery 1996). The definition of a "large" river would here be a river with sufficient energy to erode flood plain banks, even when large trees provide root reinforcement of the soil (Agee 1988). In this study, no site appears to meet this definition, at least to the degree of other sites in the literature such as the Queets River in western Washington (Abbe and Montgomery 1996). Specific references to fire and stand types in riparian forests along smaller streams are rare for pre-settlement forests, perhaps indicating that there were no widespread or dramatic differences between riparian forests and adjacent upland forests.

Within the study area in the late 19th century, patchy riparian forests were only described along the lower Skagit River. Ayres (1899) described the flood plain of the Skagit River below Lyman as "a dense tangle of alder, vine maple, crab apple, devil's club, and ferns, among which are growing large trees of Sitka spruce, cedar, large-leaved maple, cottonwood, and, occasionally, silver fir." Such a description suggests that fire was perhaps less common than in the surrounding Douglas fir forests, but also that channel migration may have created forest patches of different ages and species composition. Because these stand types were restricted to a small portion of the study area (covering only the extreme lower portions of Hansen and Red Cabin Creek sub-basins), I assumed that estimates of fire recurrence intervals presented in this chapter were suitable for all of the sub-basins in the study area. This assumption would not apply to assessments of LWD recruitment along larger rivers such as the lower Suiattle and lower Skagit Rivers.

Natural variation in stand age across a landscape can be crudely estimated in two ways. First, for areas where fires were typically stand-replacing fires and the distribution of fire inter-arrival times has been estimated, one can reconstruct stand ages by modeling the random of occurrence of fires over a long time period. The distribution of fire inter-arrival times is often estimated based on the distribution of forest ages (e.g., Booth 1991, Agee 1988). Fire frequencies have been estimated at numerous locations in the United States (Heyerdahl et al. 1995). However, estimates of fire frequency on upslope areas may not be suitable for riparian forests because fire inter-arrival times may have been substantially longer on flood plains of large rivers, and other stand-modifying processes are active there (Agee 1988). A second approach is to use historic information on stand types, timber volume, or stand ages to estimate dominant conditions on the landscape prior to logging or other land uses. Both methods describe the patchy distribution of forest ages on the landscape prior to wide spread logging.

This appendix describes variation in forest ages and selected stand attributes in the study area prior to European settlement. Forested areas were divided into two elevation zones based on Ayres (1899) and Franklin and Dyrness (1979). These zones are the western hemlock zone (up to approximately 700 m elevation) and the Pacific silver fir zone (approximately 500 to 1500 m elevation). Higher elevation areas were not considered in the analysis, primarily because forests are sparse and little land use occurs at elevations above 1500 to 2000 m. Fire recurrence intervals and stand age distributions were estimated separately for each zone.

Fire regime in the western hemlock zone

The distribution of times between stand-replacing fires in the western hemlock zone is difficult to estimate from literature, primarily because some fires in the western hemlock zone were ground fires that left most large trees alive (Heyerdahl et al. 1995). For all fires including ground fires, the return interval for western hemlock-Douglas fir forests has been estimated at 96 years with a standard deviation of 22 years (number of sites = 13, data from Heyerdahl et al. 1995). However, this return interval does not represent the recurrence interval of stand-replacing fires, and cannot be used to generate a probability distribution of fire inter-arrival times for stand-replacing fires, or to model stand ages. Agee (1993) concluded that an overall mean recurrence interval for fires in Douglas fir and western hemlock stands is about 230 years with wide variation about the mean. However, he also suggested that the mean may not be a good representation of recurrence intervals of stand replacing fires at any one location.

In the absence of applicable regional data for the western hemlock and Douglas fir zone, I sought out fire history information specific to the study area. Fire history maps of the Sauk River basin were produced for the Sauk River and Sauk River Forks Watershed Analysis (USDA Forest Service 1996a). These maps display the REAP fire history results, indicating that three large stand-replacing fires occurred in the area during the last several centuries (1508 AD, 1701 AD, and 1834 AD). Only the last fire has the possibility of having been set by white settlers, and this possibility seems remote given that settlement of this area was not significant until the late 1800s. I therefore assumed that all three fires were natural. There also remains the possibility that these fires were set by Native Americans.

Areas of stands in the western hemlock zone were estimated by counting grid points overlain on the fire map. In the year 1900, the distribution of stand ages (i.e., time since last burned) was as shown in Figure A-1. In that year, nearly 70% of the stands in the western hemlock zone would have been more than 199 years old, and no stands would have been less than 66 years old. Immediately after the most recent fire (ca. 1834), 32% of the stands would have been recently burned (assumed here to be age 0), and the remaining 68% of the stands would have been $\geq$ 133 years old. The average interval between fires can be estimate from these data using the equation $\Sigma f_t = 1 - e^{-pt}$ (Booth 1991). The eqaution assumes a negative exponential distribution of stand ages, where f_t is the frequency of stands of age t and p is the annual probability of fire at point. By this calculation the estimated probability of burning in any one year (p) ranges from 0.0044 to 0.0056 (for stand age distributions in the years 1900 and 1834, respectively), and the average time between fires ($1/p$) ranges from 179 years to 227 years.

These results are consistent with Gannett's (1899) description of forests west of the Cascade range and in Skagit County. Gannett (1899) described forests west of the Cascade range as

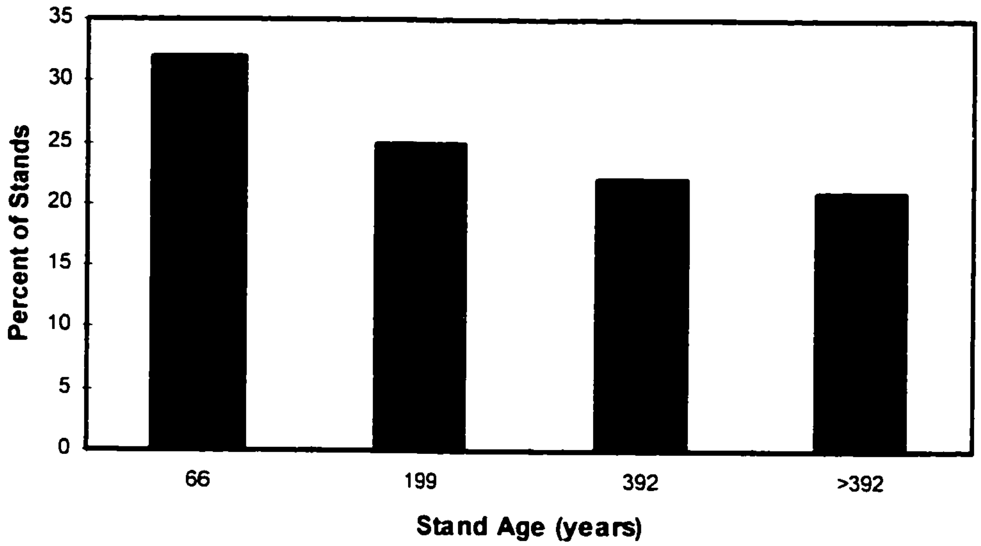


Figure A-1. Distribution of stand ages (i.e., time since last fire) in western hemlock zone of the Sauk River basin, as of the year 1900 AD. Data from Sauk River and Sauk River Forks Watershed Analysis (USDA Forest Service 1996a).

consisting of 64% fir, 16% cedar, 14% hemlock and 6% spruce. Because Douglas fir rarely lives beyond about 400 years (Munger 1940), the majority of forests must have been less than 400 years old. In the Sauk River example, about 80% of the forest area was less than 400 years old during the six decades prior to 1900, and the median age of stands was between 133 and 199 years. Furthermore, records of diameter at the stump and tree ring counts found in Ayres (1899) indicate that diameters of the largest Douglas fir at ages 133 yr and 199 yr would be approximately 1.00 m (39 in) and 1.36 m (52 in), respectively (Figure A-2). These estimated diameters are also consistent with a description of marketable timber extracted from Douglas fir forests at that time, which were logs "that will square 15 inches, or 22 inches in diameter" and 32 feet long (Gannet 1899).

Fire regime in the Pacific silver fir zone

In silver fir forests (approximately 500 to 2000 m elevation), most fires are severe crown fires that kill most trees in the stand (Heyerdahl et al. 1995). Hence, it is reasonable to assume that the distribution of times between fires represents the distribution of stand replacing fires, and that this distribution of times between fires can be used to estimate the distribution of forest ages in the absence of land use. The distribution of times between fires in silver fir in Washington state appears to be skewed toward higher ages (data from Heyerdahl et al. 1995), so I used a log-normal distribution to represent the distribution of stand ages. The geometric mean interval between fires is 380 years (n=10). With this distribution I modeled a sequence of 30 fires for 100 sites where each of the intervals between fires was randomly selected from the log-normal distribution. Stand age was then calculated for each of the 100 sites at t = 5000 years to generate a frequency distribution of stand ages at a point in time (Figure A-3). The result indicates that one would expect wide variability in stand ages in unmanaged silver fir forests, with a median age of 225 years. An alternative calculation describes the stand age distribution by a least squares fit of the equation $\Sigma f_t = 1 - e^{-pt}$ (Booth 1991). This approach yields a median stand age of about 260 years.

In the silver fir zone, Ayres (1899) estimated merchantable timber volume per acre to be very low by the standards of the time, which "covered the red fir that will square 15 inches, or 22 inches in diameter, and such spruce, cedar, and hemlock as may be growing among it of similar size and good quality" (p. 304, Ayers 1899). However, using eastern standards, which included

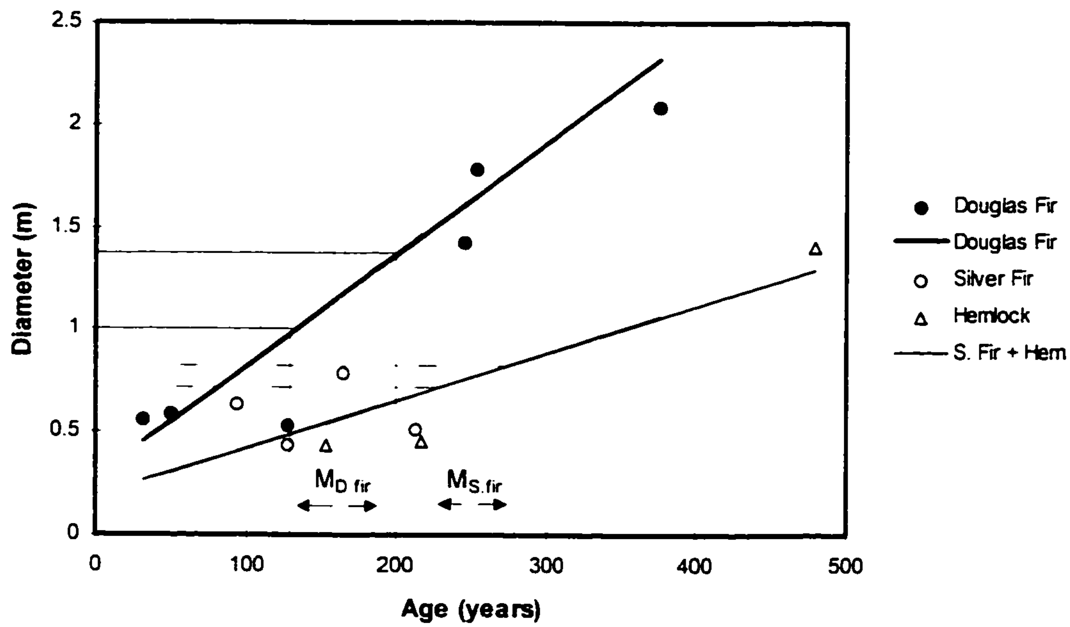


Figure A-2. Diameter at the stump plotted against age for Douglas fir (*Pseudotsuga menziesii*), and for Pacific silver fir (*Abies amabilis*) and western hemlock (*Tsuga heterophylla*). Data from Ayres (1899). Stumps heights vary from 2 to 12 feet (0.6 m to 3.7 m). Light line is the regression for western hemlock and Pacific silver fir above 500 m elevation: diameter = 0.19 + 0.0023(age); $r^2 = 0.71$, $p = 0.017$. Estimates of median age of Pacific Silver fir and western hemlock stands ($M_{S\ fir}$) prior to European settlement ranged from 225 to 260 years. Diameters (dbh) of largest silver firs or hemlocks in those stands would be expected to range from about 0.67 m (26 in) to 0.77 m (30 in). Heavy line is the regression for Douglas fir: diameter = 0.28 + 0.0049(age); $r^2 = 0.90$, $p = 0.004$. Estimates of median age of Douglas fir stands ($M_{D\ fir}$) prior to European settlement ranged from 133 to 199 years. Diameters (dbh) of largest Douglas firs in those stands would be expected to range from about 1.00 m (37 in) to 1.36 m (49 in).

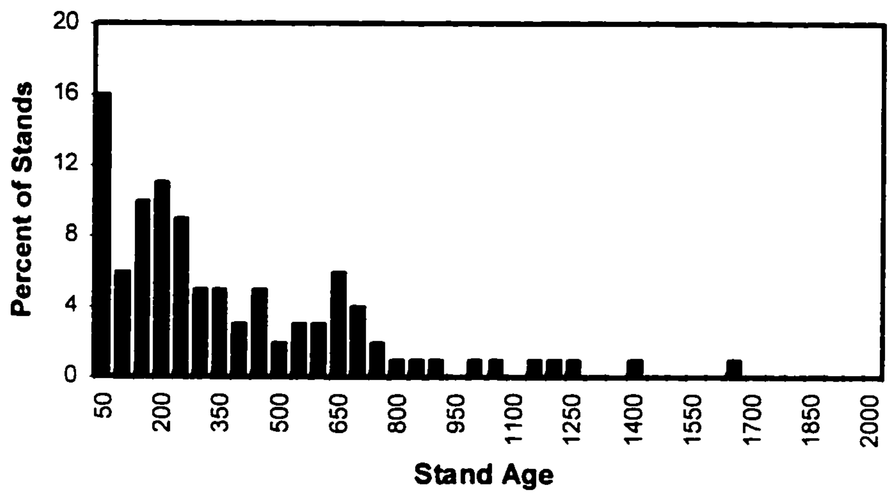


Figure A-3. Distribution of stand ages after 5000 years of randomly generated fires in 100 silver fir sites. Times between fires were based on a log-normal distribution with geometric mean of 380 years. Data from Heyerdahl et al. (1995).

"all timber that would make a log 8 inches at the small end, 12 feet long, and would scale two-thirds of a full scale" (p. 304, Ayers 1899), Ayres estimated a significant volume of timber in the reserve. By his account, a representative acre of western hemlock and silver fir contained 40 trees ranging from 10 to 30 inches in diameter (88 stems/ha from 25 to 75 cm diameter). Approximately 50 smaller stems (4 to 10 inches diameter) were also found in the 1 acre plot (198 stems/ha from 10 to 25 cm diameter). Of the larger trees, about 1/3 were hemlock and the remainder silver fir (p. 303).

Based on data in Figure A-2, trees in the silver fir zone with a diameter of 75 cm would be approximately 257 years old, which is consistent with the median age estimated from the fire recurrence interval (225 to 260 years). However, the regression is strongly influenced by a single 479 year-old hemlock at an elevation of 750 m. The regression for Pacific silver fir alone was not significant ($n = 3$). Nevertheless, the average diameter of three silver fir with a mean age of 168 years was 58 cm, suggesting that trees 75 cm in diameter would be > 168 years old. All of these analyses are consistent with a median stand age in the silver zone between 225 and 260 years.

Appendix B. Modeling riparian forest growth and LWD recruitment.

Modeling of riparian forest growth and LWD recruitment requires the use of two separate models. The inputs to each model, the linkages between them, and the outputs are shown in Figure 5-5 (p.101). This appendix first describes the forest growth modeling, focusing on the calibration of Forest Vegetation Simulator (FVS) for each riparian species considered in this dissertation. It then briefly describes the structure of Riparian-in-a-Box II (RIAB), and describes the elements developed specifically for this dissertation. It describes how RIAB estimates recruitment and depletion of LWD, and how RIAB estimates the minimum size of pool-forming LWD. Finally, it describes the sensitivity of FVS and RIAB to various inputs to the models.

Calibration of growth and yield model

The growth and yield model (FVS), which simulates forest growth and mortality, is separate from the recruitment model (Riparian-in-a-Box), which calculates LWD recruitment and pool-riffle characteristics. FVS "grows" trees in the riparian forest and describes attributes of the stand at each time step. It is a multi-species model that includes growth algorithms for Douglas fir, western hemlock, western red cedar, red alder, black cottonwood, big leaf maple and 31 other species. It simulates the growth of individual trees in the stand, including a stochastic representation of the variability among individual trees. The model has tremendous flexibility, and thus does not generally predict tree growth accurately without calibration.

I calibrated the model for four species considered in this dissertation: Douglas fir, western hemlock, western red cedar, and red alder. For each species, I calibrated growth to local riparian data where possible (Douglas fir, western hemlock, red alder), and to literature data where local riparian data where insufficient (western red cedar). For each species I first calibrated the basal area growth increment (the annual or decadal change in cross sectional area of the tree at 4.5 feet above ground), from which FVS derives diameter at breast height (the diameter of the tree at 4.5 feet above ground) and tree height. For Douglas fir and western hemlock I had only maximum diameters from field data for calibration, and I assumed that the diameter distribution in the stand was approximately correct when maximum diameters were similar. For western red cedar and red alder I had a range of diameter measurements in the field data, and I calibrated quadratic mean diameter from the model to the middle of the range from field data. Once diameter growth approximately matched the field data, I calibrated height growth. Height growth calibrations were similar to the diameter calibrations in the types of field data available and calibration procedure. Finally, I calibrated mortality for conifer based on the stand density index (SDI):

$$SDI = tpa \times \left(\frac{dbh}{10}\right)^{x}$$

where *tpa* is the number of trees per acre in the stand, *dbh* is diameter at breast height, and x is a species-specific coefficient. I adjusted mortality rates such that SDI for Douglas fir was generally less than 500 (Jeff Welty, forest biometrician, Weyerhaeuser Corporation, personal communication), SDI for western hemlock was generally less than 485 (Meyer 1937), and SDI for red cedar was generally less than 550 (based on Smith 1987). For red alder, I calibrated mortality based on numbers of trees per acre in red alder plots in the study area.

Final calibrated diameter and height curves for each species are shown in Figures B-1 through B-4. To facilitate comparisons in growth patterns among species, Figure B-5 shows the calibrated curves for diameter, height, and trees per acre for all species.

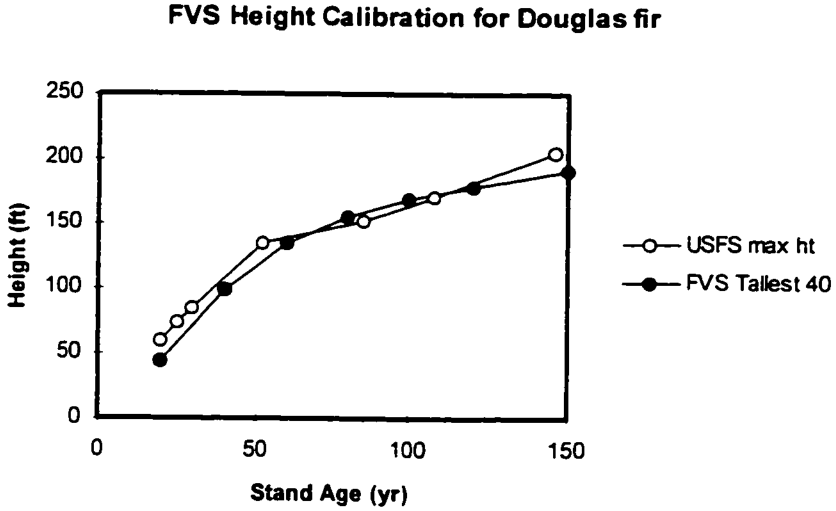
LWD recruitment and depletion

The growth model described in the preceding section estimates mortality in the developing stand, and predicts the number and size of trees that will die over time. Riparian-in-a-Box then predicts how many of these fallen trees will enter the stream assuming that fall direction is random, and assuming that all dead trees fall (Kennard et al. In press). The model calculates the proportion of fallen trees that will reach the stream based on Van Sickle and Gregory (1990), who described a method for calculating the probability of a fallen tree reaching a stream (P_s) based on the length of the fallen tree (h) and distance to the stream (z). Their equation is

$$P_s = \cos^{-1}(z/h)/\pi.$$

Trees falling parallel to or away from a stream bank obviously have no chance of reaching the stream, and trees falling toward the stream have increasing probability of entering the stream as the fall direction varies from nearly parallel to the channel, to perpendicular to the channel. The size of the bole reaching the stream is then calculated based on tree taper, the length of the fallen tree, and distance to the stream for a given fall direction.

Depletion rate of conifer LWD decreases with increasing size of LWD (Murphy and Koski 1989), and may also vary with LWD abundance. Across all sizes of conifer LWD, depletion rate averages between 1.5% and 2% per year. Hardwood LWD is generally less decay resistant than



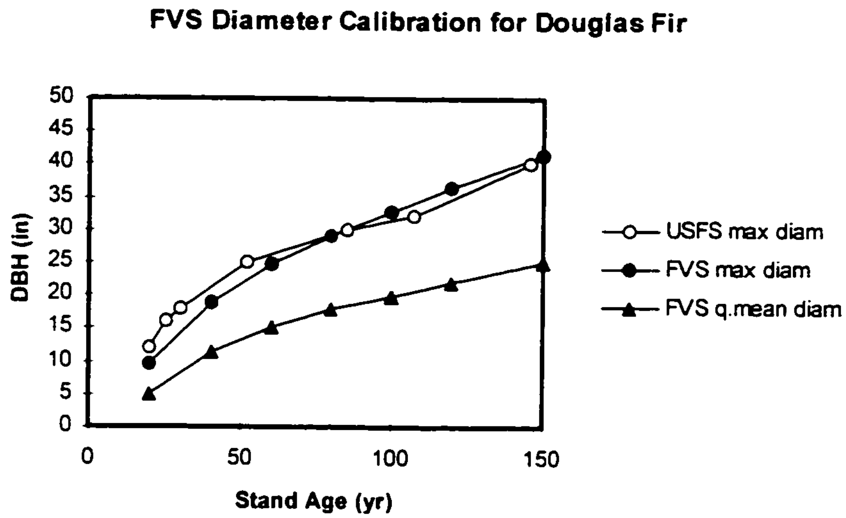
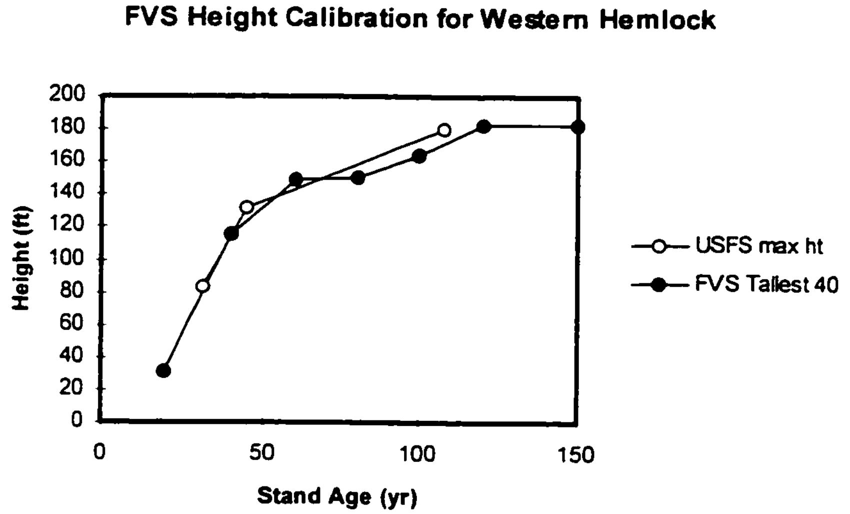


Figure B-1. FVS calibration results for Douglas fir for height growth (top) and diameter growth (bottom). Field data from Jan Henderson, plant ecologist, USFS Mt. Baker Snoqualmie National Forest, unpublished data.



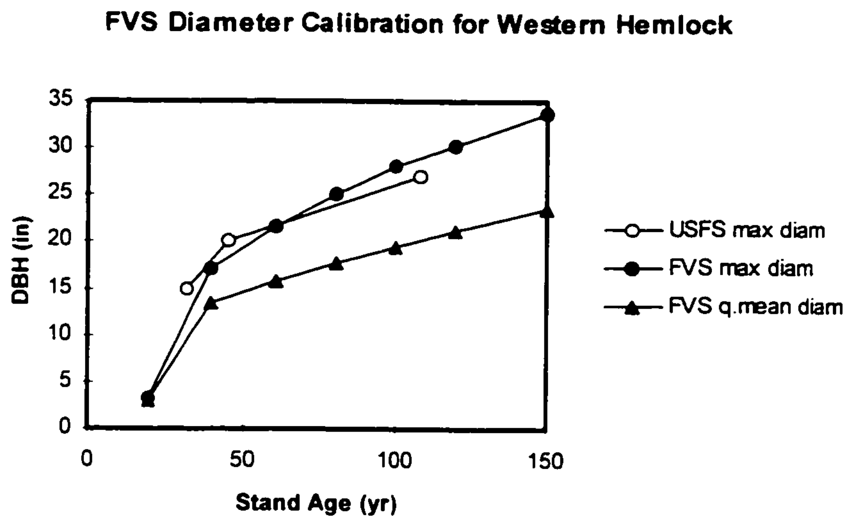
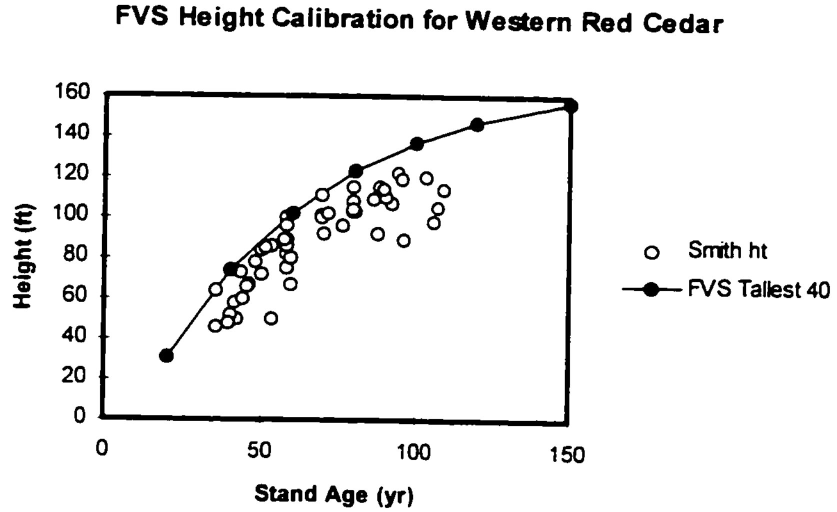


Figure B-2. FVS calibration results for western hemlock for height growth (top) and diameter growth (bottom). Field data from Jan Henderson, plant ecologist, USFS Mt. Baker Snoqualmie National Forest, unpublished data.



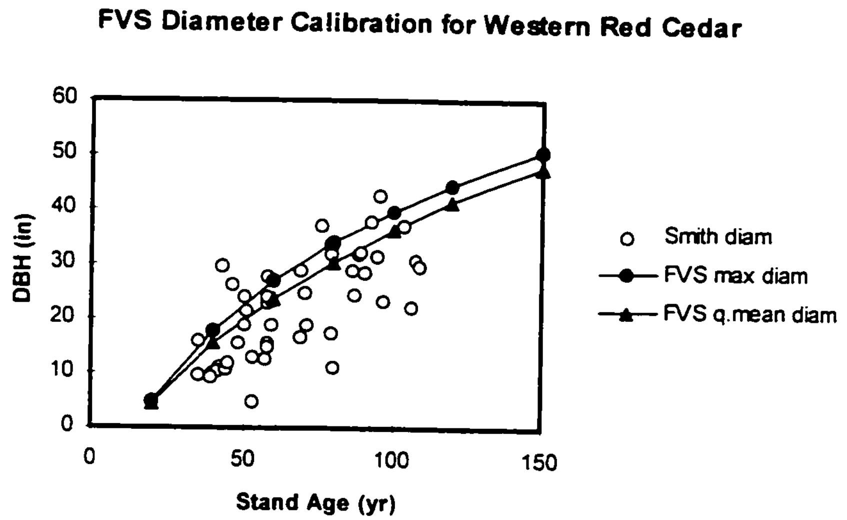
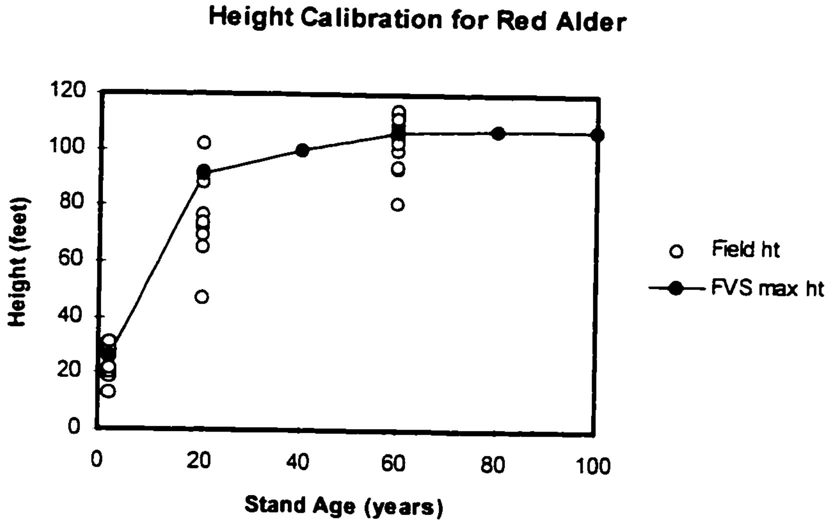


Figure B-3. FVS calibration results for western red cedar for height growth (top) and diameter growth (bottom). Cedar data from Smith (1987).



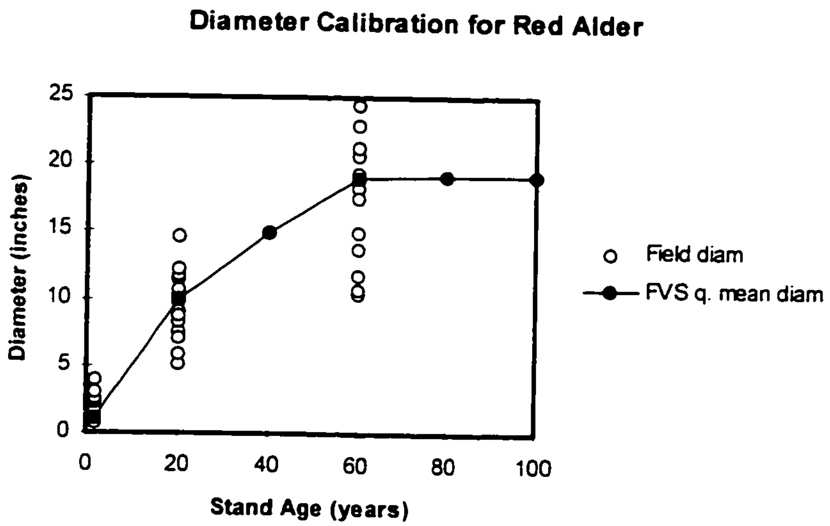


Figure B-4. FVS calibration results for red alder for height growth (top) and diameter growth (bottom). Field data from this dissertation.

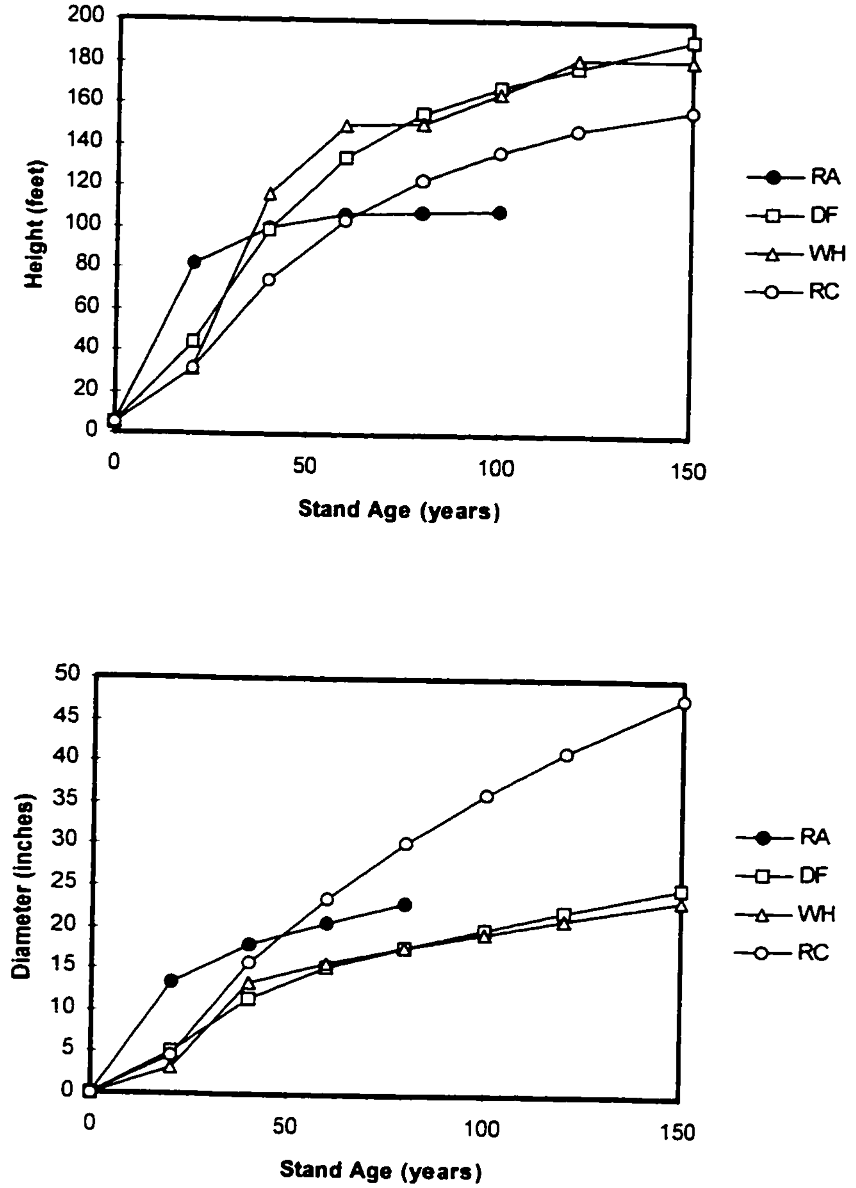


Figure B-5. Plots of FVS predicted height and diameter growth after calibration: Douglas fir (DF), western hemlock (WH), western red cedar (RC), and red alder (RA).

conifer LWD (e.g., Scheffer and Cowling 1966), and so has a higher depletion rate. Based on data in Andrus et al. (1988), recruitment of red alder ceases after about 70 years, and alder LWD abundance in the channel decreases to near zero over a period of about 70 years. I estimated hardwood depletion per year from these data using the equation

$$LWD_t = LWD_0*(1-d_{annual})^t$$

where LWD_t is LWD abundance at time t, LWD_0 is initial LWD abundance, d_{annual} is the annual depletion rate, and t is elapsed time in years. With $LWD_0 = 1.25\ m^3/100m^2$, $t = 70$ years, and $LWD_t = 0.1\ m^3/100m^2$, the calculation yielded a depletion rate of about 3.5% per year for hardwoods.

These two depletion rates (1.5% per year for conifer and 3.5% per year for hardwood) are average depletion rates that encompass different decay rates (a function of species) as well as different export rates (a function of channel size and LWD size). Murphy and Koski (1989) showed that depletion rate of small diameter conifer LWD (typically between 1.5% and 3% per year for LWD <60 cm diameter) is greater than the depletion rate of large diameter conifer LWD (typically less than 1.5% per year for LWD >60 cm diameter). The smallest diameter LWD (<30 cm) typically has a depletion rate between 2% and 3% per year, which is fairly close to the average hardwood depletion rate. This suggests that depletion rates for small diameter conifer and hardwood LWD should both be between 2% and 3.5%, and that the difference in average depletion rates between hardwood and conifer may be primarily a function of the smaller average diameter of hardwood LWD.

Differences in decay rates among several conifer species and red alder appear to be insignificant based on above ground durability tests. Sapwood durability of red cedar, Douglas fir, Engelmann spruce, and red alder all ranged from 11 to 16 years in the above ground tests, and heartwood durability exceeded 30 years for Douglas fir and red alder, but was only 14 years for Engelmann spruce (Highley 1995). Because, these data do not suggest significant differences between red alder and Douglas fir (the two most common species in the study area), they indirectly support an assumption that depletion rates are primarily a function of LWD size. Based on this assumption, one would expect the depletion rates of both red alder and Douglas fir to be similar during the first several decades of riparian forest recovery because most recruited LWD during the first 100 years is less than 60 cm diameter (Figure B-5). Thus, I modeled all species the same average depletion rate. Because the model does not account for breakage of pieces, I modeled all of the scenarios with an average annual depletion rate of 1.5% as in Kennard et al. (In press), which is somewhat less than the previously estimated depletion rate of 2% to 3.5%.

Based on the preceding recruitment and depletion methods, Riparian-in-a-Box tallies recruited LWD over time, and also keeps track of the depletion of LWD during the model run. The net amount of LWD in the reach at any point in the model run is calculated as

$$LWD_t = LWD_0 + LWD_{recruitment} - LWD_{depletion}$$

where LWD_t is LWD abundance at time *t*, LWD_0 is initial LWD abundance, $LWD_{recruitment}$ is the amount of LWD recruited between time 0 and time *t*, and $LWD_{depletion}$ is the amount of LWD lost between time 0 and time *t*.

Estimation of minimum size of pool-forming LWD

There are several ways to quantify the size of LWD that forms pools in channels, including the average size of stable LWD (Bilby and Ward 1989) and the minimum size of pool-forming LWD (Beechie and Sibley 1997). Both values are a function of channel width. The average size of stable LWD is approximately the same as the median size of pool-forming LWD in second growth channels, which means that use of the average in the RIAB model would neglect smaller pieces that form pools. Therefore, I focused on the minimum size of LWD that forms pools.

To generate an equation describing the minimum functional size of LWD, I used two separate approaches. First, Beechie and Sibley 1997 showed that minimum size of pool-forming LWD ($diameter_{pf}$) was related to channel width (w_{bkf}) by the equation

$$diameter_{pf} \cong 28(w_{bkf})$$

where pool-forming diameter is in centimeters and channel width is in meters. Second, I developed a relationship between channel width and the minimum size of LWD that RIAB would tally. This relationship was necessary because the model does not vary depletion rates by LWD size or stream size, and all size streams would have the same estimated LWD abundance if the model tallied all LWD >10 cm diameter for all channel widths. However, we know that LWD abundance decreases logarithmically with channel width (Bilby and Ward 1989). This logarithmic relationship appears to be explained by the increased mobility of small LWD in larger channels, which comprise a decreasing proportion of the total LWD abundance as channel width increased (Bilby and Ward 1989).

To develop the relationship between channel width and a minimum LWD diameter for RIAB to tally, I assumed that Bilby and Ward (1989) LWD size data were log-normally distributed, with more pieces near the small end of the size range. I also assumed that the distribution could be reasonably represented by size classes ranging from 10 cm to 2 m in diameter, and that the distribution could be described the function $1-(1-e^{-pt})$ (Figure B-6). In this function, $1/p$ is the average diameter, which is assumed to be approximately equal to the geometric mean value in Bilby and Ward (1989). Therefore, the entire distribution of piece sizes in the smallest channels (4 m wide) can be represented by a log-normal distribution with an average diameter ($1/p$) of 35 cm (from Bilby and Ward (1989) regression). However, our distributions start at 10 cm rather than at zero, so the calculations are made with 25 cm as the average and 0.1 cm as the smallest diameter in the distribution. This gives a frequency distribution of piece sizes ranging from 0.1 to 190 cm, which is then plotted as a distribution ranging from 10 cm to 200 cm.

We know from Bilby and Ward (1989) that LWD frequency goes from 0.6 LWD/m in a 4 meter wide channel to 0.1 LWD/m in a 20 m wide channel. I assume that the size distribution of LWD inputs for any size channel are represented by that of a 4 m wide channel, and that the ultimate size distribution at any channel width is a function of the export of smaller LWD. That is, I assume that smaller pieces are exported from larger reaches, and that the upper size limit of exported LWD is related to channel width. To mimic these relationships in a recruitment model, I developed a linear function that described the smallest stable piece of LWD as a function of channel size. Graphically, the number of pieces in the distribution is 0.6 when all pieces greater than 10 cm are included. However, to mimic the 0.1 LWD/m seen in a 20 m wide channel, we count only LWD >50 cm diameter (Figure B-6). Thus, minimum LWD diameter in the tally is

$$(\text{minimum diameter}) = 2.5(w_{bkf}) \qquad (1)$$

where w_{bkf} is in meters and diameter is in centimeters. This is similar to minimum functional diameter in Beechie and Sibley (1997):

$$(\text{minimum diameter}) \cong 2.8(w_{bkf}) \qquad (2)$$

The equation from Beechie and Sibley (1997) in essence represents the average minimum from a number of channels, rather than the lower limit defined by the data. Therefore, I use equation 1, which more closely represents the lower limit of the data in Beechie and Sibley (1997) and retains a greater number of the pieces in the tally. For channels 4 m wide, equation 1 counts

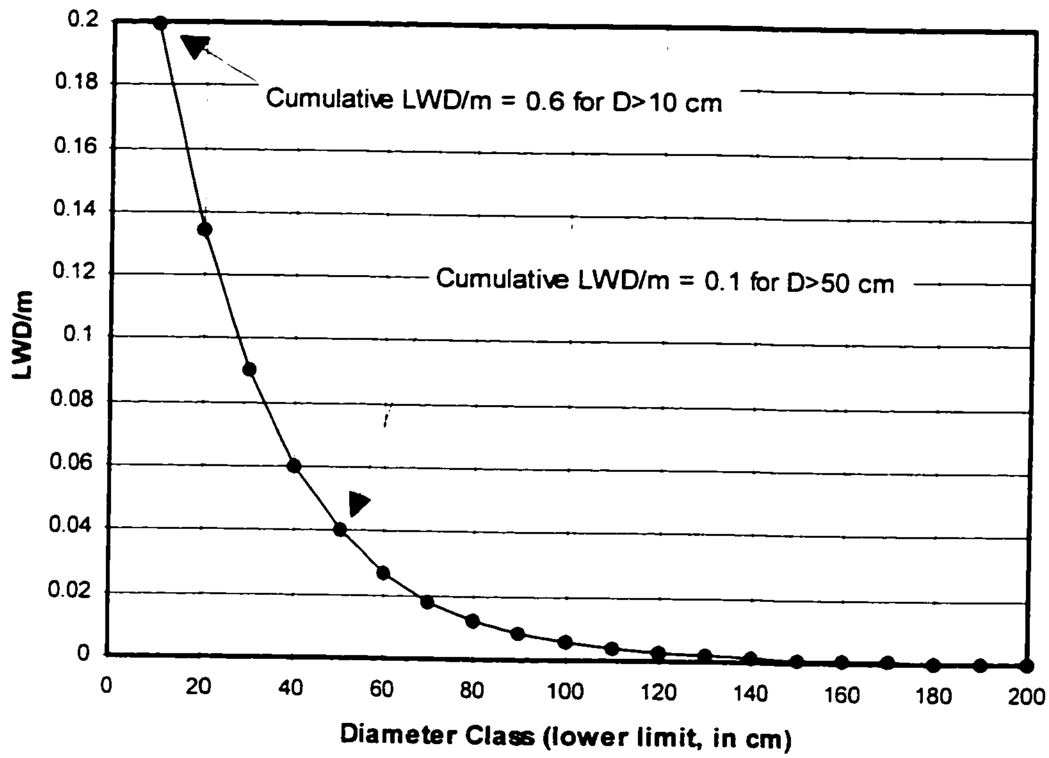


Figure B-6. Graph of hypothetical log-normal distribution of LWD sizes in a stream channel based on Bilby and Ward (1989). For any channel width, the model estimates the minimum size of LWD to count using the equation (minimum diameter) = 2.5(w_{bkf}). For example, in a 4 m wide channel LWD abundance is estimated at about 0.6 pieces per meter, which includes all pieces greater than 10 cm diameter. For a larger channel (20 m wide) the model estimates abundance at about 0.1 piece per meter because it counts only LWD > 50 cm diameter. Results are consistent with the relationship between LWD/m and channel width described by Bilby and Ward (1989).

pieces > 10 cm and equation 2 counts pieces > 12 cm. For channels 20 m wide, equation 1 counts pieces > 50 cm and equation 2 counts pieces > 57 cm.

The result of this function is that the total abundance of LWD in a reach predicted by the model is function of channel width similar to that shown by Bilby and Ward (1989) (Figure B-7). Thus, in a small channel (4 m wide) LWD abundance is estimated at about 0.6 pieces per meter, which included all pieces greater than 10 cm diameter. For a larger channel (20 m wide) the model estimates abundance at about 0.1 piece per meter because it counts only LWD > 50 cm diameter. Both results are consistent with the Bilby and Ward (1989) relationship. Thus, total LWD abundance estimated by the model will more closely approximate measured LWD abundance as a function of channel width, and equations that estimate pool abundance from LWD abundance will be more accurate.

Model accuracy and sensitivity

Model accuracy

Based on comparison to one data set from Oregon (in Andrus et al. 1988), the FVS and RIAB models predict the temporal pattern of hardwood and conifer LWD abundance in a stream channel with reasonable accuracy (Figure B-8). The Oregon data show abundance of LWD originating from hardwood stands increasing after about 20 years and reaching a maximum at about 70 or 80 years after the stand was established. Model predictions for a channel of similar width (about 7.5 m) show a similar pattern, with LWD abundance from a red alder stand beginning after about 10 years and reaching a maximum at 70 to 80 years. Conifer debris in the Oregon streams increased slightly after about 20 years, then began to increase rapidly after about 50 years and continued increasing until at least 140 years after disturbance. The models predict approximately the same pattern, with LWD abundance from a Douglas fir stand increasing rapidly after about 40 years and continuing to increase at least out to 100 years.

Sensitivity analysis

An important consideration in any use of these models is the sensitivity of LWD recruitment to changes in input variables for either Forest Vegetation Simulator (FVS) or Riparian-in-a-Box II (RIAB). This section first describes some of the sensitivities of FVS to changes in site characteristics, which have been extensively discussed in literature documenting the model

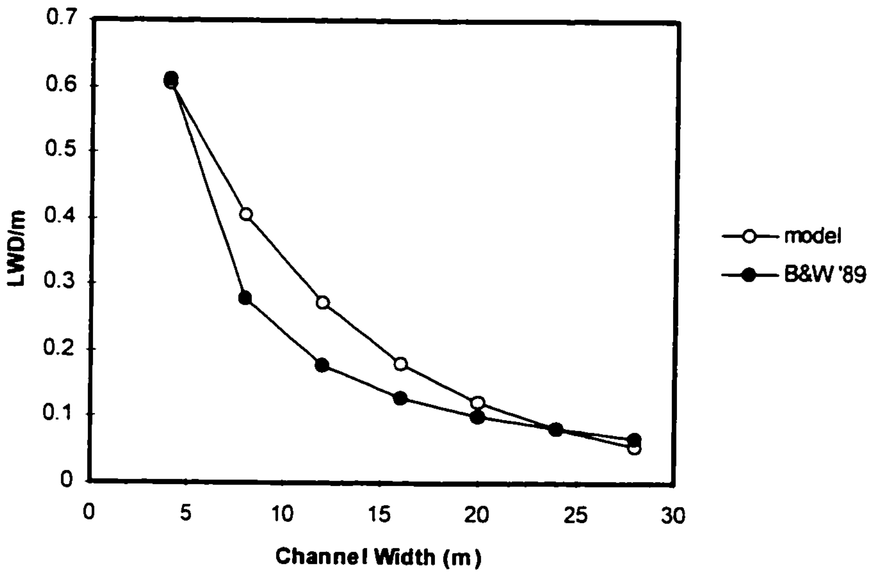


Figure B-7. Comparison between the number of pieces of LWD that the Riparian-in-a-Box II model will tally (open circles), and the field data from Bilby and Ward (1989).

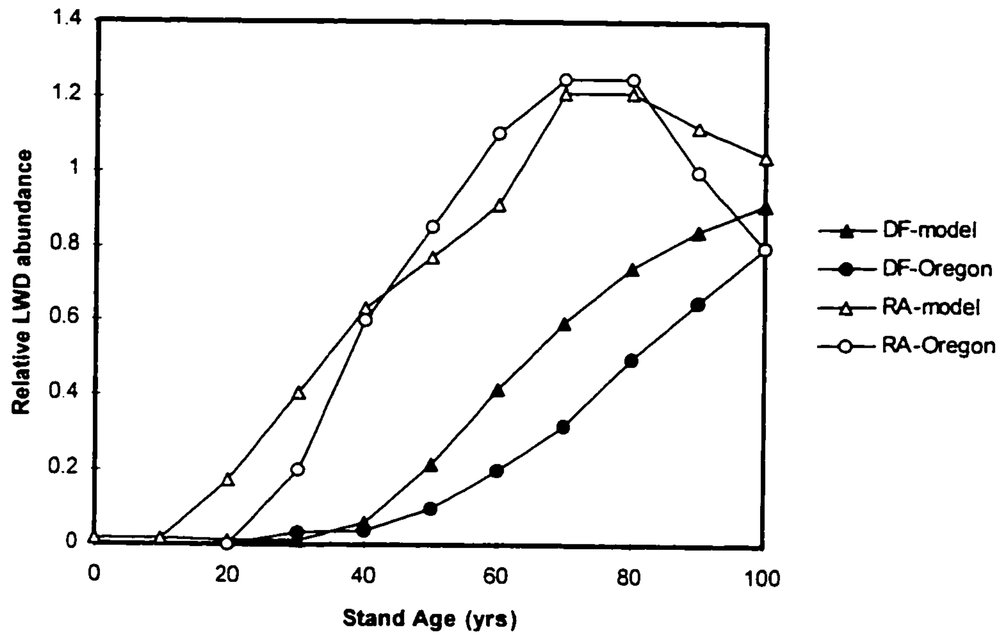


Figure B-8. Comparison of predicted LWD abundance (from FVS and RIAB model simulations) to empirical LWD abundance in Oregon streams (adapted from Andrus et al. 1988) for hardwood (RA) and conifer (DF). Model runs were based on a channel width of 7.5 m, which is similar to the channel widths included in the data from Andrus et al. (1988).

structure and usage (e.g., Wykoff et al.1982). This section then discusses the sensitivity of LWD recruitment outputs to changes in the two most important input options in RIAB: site class and depletion rate.

FVS is sensitive to a variety of site parameters including aspect, slope, and elevation (Wykoff et al.1982). These sensitivities vary by species, and interact with each other to create complex growth responses to site characteristics. Fortunately, these sensitivities are mostly quite small for a given site index and stocking, indicating that they will not have large effects on the LWD recruitment modeling. This section does not attempt to give a complete summary of these sensitivities, but only to illustrate their approximate magnitudes for two species occurring in the study area, Douglas fir and western red cedar.

Where riparian manipulations are intended to benefit anadromous salmonids, it is appropriate to consider the sensitivity of tree growth to variation in the elevation of riparian forests, which range from sea level to an elevation of 700 m in the study area (Beechie et al.1994). Growth of Douglas fir and red cedar are both relatively insensitive to elevation over such a range, although the elevations evaluated in the original model were higher than those in the study area (ranging from about 610 m to 2440 m, or 2000 feet to 6000 feet). Based on the original sensitivity analysis in Wykoff et al. (1982), Douglas fir diameter growth varies by only $\pm$2% over an elevation range of 700 m, and red cedar diameter growth varies by $\pm$6%.

Slope and aspect also influence growth rates of these two species, and the sensitivity of each is dependent on the value of the other. For example, western red cedar diameter growth rate is relatively insensitive to slope on a south aspect ($\pm$1.5% over the slope range of 0.0 to 0.8), but much more sensitive on a north aspect ($\pm$7% over the same slope range). Douglas fir is more sensitive to slope than red cedar, with diameter growth varying by +11% over slopes from 0.0 to 0.8 on a north aspect, and $\pm$12% over slopes from 0.0 to 0.8 on a south aspect. Sensitivity to aspect for both Douglas fir and western red cedar is essentially zero on flat ground, but increases to about $\pm$5% for Douglas fir and $\pm$10% for western red cedar when slope is about 75%.

I assessed sensitivity to input parameters in Riparian-in-a-Box by comparing LWD abundance at 100 years for varying levels of site class and depletion. The sensitivity analysis was based on Douglas fir regeneration at an elevation of 100 feet (30 m), and on a low-slope (2%) site with a south aspect. The channel characteristics for the analysis were a slope of 0.5%, a channel width of 10 m, and an initial LWD loading of 0.01 LWD/m. The low initial LWD loading (required to be a non-zero number in the model) focuses the analysis on LWD that is recruited from the new stand, and avoids the confusion of trying to distinguish new LWD from residual (old stand) LWD.

The base depletion rate for the analysis is 2% per year, and depletion was varied from 0.5% per year (-75%) to 3.5% per year (+75%) based on the variability in depletion rates calculated by Murphy and Koski (1989). A 75% increase in depletion rate results in a 25% decrease in LWD/m at 100 years (Figure B-9). A 75% decrease in depletion rate results in a 50% increase in LWD/m after 100 years. The base site index at 100 years was 170 feet, and site index was varied from 140 feet (-18%) to 210 feet (+24%) based on the range of site index values in the study area (Jan Henderson, forest ecologist, USFS, unpublished data). A 24% increase in depletion rate results in a 19% decrease in LWD/m at 100 years, and an 18% decrease in depletion rate results in a 19% increase in LWD/m after 100 years. Thus, LWD/m is more sensitive to site index than to depletion rate based on the greater change in LWD/m per unit change in the input parameter. However, because there is greater overall variation in depletion rates, there is a greater overall range of LWD/m at 100 years due to depletion rate than there is due to site index.

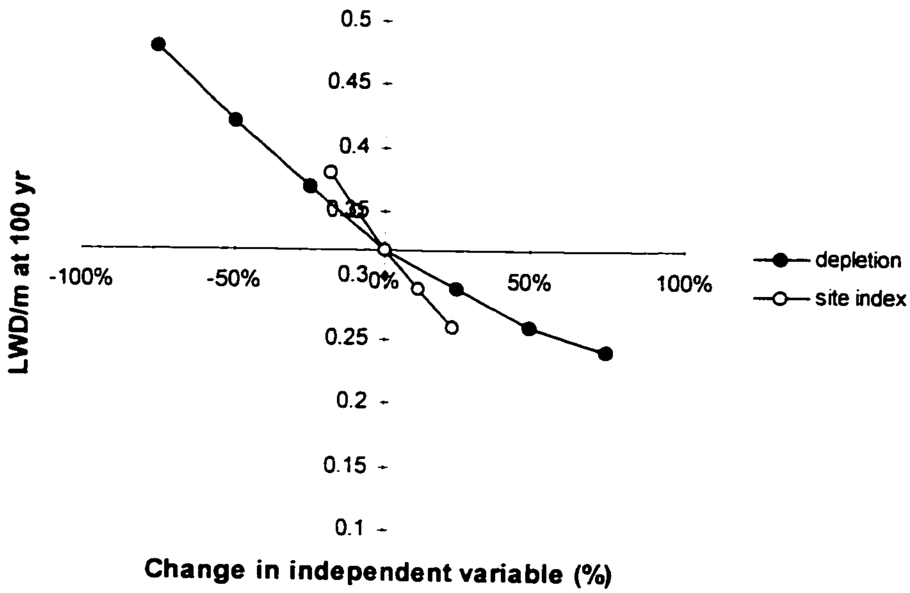


Figure B-9. Sensitivity of predicted LWD recruitment from a Douglas fir stand to changes in depletion rate and site index. The analysis compares LWD/m at 100 years for varying levels of inputs. LWD depletion rate from a stream channel was varied in the Riparian-in-a-Box model, and site index was varied for Douglas fir using DFSIM (Jeff Welty, Weyerhaeuser Company, unpublished data). The base depletion rate for the analysis is 2% per year, and depletion was varied from 0.5% per year (-75%) to 3.5% per year (+75%) based on the variability in depletion rates calculated by Murphy and Koski (1989). The base site index at 100 years was 170 feet, and site index was varied from 140 feet (-18%) to 210 feet (+24%) based on the range of site index values in the study area (Jan Henderson, forest ecologist, USFS, unpublished data). Site and channel characteristics are described in text.

Vita

Timothy J. Beechie

Ph.D., College of Forest Resources, University of Washington, Seattle. 1998.
Dissertation title: Rates and pathways of recovery for sediment supply and woody debris recruitment in northwestern Washington streams, and implications for salmonid habitat restoration.

M.S., School of Fisheries, University of Washington, Seattle. 1990.
Thesis title: Evaluation of TFW Stream Classification System: Stratification of physical habitat area and distribution.

B.S., Department of Geological Sciences, University of Washington, Seattle. 1983.

Other publications

Beechie, T., E. Beamer, B. Collins, and L. Benda. 1996. Restoration of habitat-forming processes in Pacific Northwest watersheds: a locally adaptable approach to salmonid habitat restoration. Pages 48-67 *In* D. L. Peterson and C. V. Klimas, eds. The Role of Restoration in Ecosystem Management. Society for Ecological Restoration, Madison, Wisconsin.

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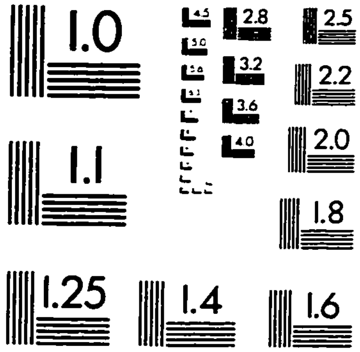
Beechie, T.J. and T.H. Sibley. 1997. Relationships between channel characteristics, woody debris, and fish habitat in northwestern Washington streams. Transactions of the American Fisheries Society 126:217-229.

IMAGE EVALUATION TEST TARGET (QA-3)



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