Chapter 9: The Aquatic Conservation Strategy of the Northwest Forest Plan: An Assessment After 10 Years

Gordon H. Reeves

Introduction

The Aquatic Conservation Strategy (ACS) of the Northwest Forest Plan (the Plan) is a regional strategy designed to restore and maintain the processes that create and maintain conditions in aquatic ecosystems over time across the area inhabited by the northern spotted owl (see appendix for species names). It seeks to prevent further degradation of aquatic ecosystems and to restore habitat and the ecological processes responsible for creating of habitat over broad landscapes, as opposed to individual projects or small watersheds (USDA and USDI 1994). The foundation of the ACS is a refinement of earlier strategies, "The Gang of Four" (Johnson and others 1991), PacFISH (USDA 1992), and the Scientific Assessment Team (Thomas and others 1993). Its primary objectives are to maintain and restore:

- The distribution, diversity, and complexity of watershed and landscape-scale features to ensure protection of the aquatic ecosystems to which species, populations, and communities are uniquely adapted.
- The spatial and temporal connectivity within and between watersheds.
- The physical integrity of aquatic ecosystems, including shorelines, banks, and bottom configurations.
- Water quality necessary to support healthy riparian, aquatic, and wetland ecosystems.
- The sediment regime under which the aquatic ecosystem evolved.
- Instream flows sufficient to create and sustain riparian, aquatic, and wetland habitats and to retain patterns of sediment, nutrient, and wood routing.
- The timing, variability, and duration of flood plain inundation and water table elevation in meadows and wetlands.
- The species composition and structural diversity of plant communities in riparian zones and wetlands.
- Habitat to support well-distributed populations of native plant, vertebrate, and invertebrate riparian-dependent species.

In the short term (10 to 20 years), the ACS was designed to protect watersheds that currently had good habitat and fish populations (FEMAT 1993). The long-term goal (100 years) was to develop a network of functioning watersheds that supported populations of fish and other aquatic and riparian-dependent organisms across the Plan area (USDA and USDI 1994).

The ACS contains four components to meet these goals and objectives:

- Watershed analysis: Watershed analysis is an analytical process to characterize watersheds and identify potential actions for addressing problems and concerns and to identify possible management options. It assembles information necessary to determining the ecological characteristics and behavior of the watershed and to develop options to guide management in the watershed, including adjusting riparian reserve boundaries.
- Riparian reserves: Riparian reserves define the outer boundaries of the riparian ecosystem. They are the portions of the watershed most tightly coupled with streams and rivers. They provide the ecological functions and processes necessary to create and maintain habitat for aquatic- and riparian-dependent organisms over time, provide

dispersal corridors for terrestrial organisms, and to provide connectivity in a watershed. The boundaries were interim until a watershed analysis was completed, at which time they could be modified depending on suggestions made in the watershed analyses.

- Key watersheds: Key watersheds are intended to serve as refugia for aquatic organisms, particularly in the short term for at-risk fish populations, to have the greatest potential for restoration, or to provide sources of high-quality water. Tier 1 key watersheds currently have good populations or habitat, a high restoration potential, or both. Tier 2 key watersheds provide sources of high-quality water.
- Watershed restoration: Watershed restoration is designed to recover degraded habitat. Restoration activities focus on restoring the key ecological processes required to create and maintain favorable environmental conditions for aquatic and riparian-dependent organisms.

The ACS also includes standards and guidelines that apply to management activities in riparian reserves and key watersheds.

The primary objective of this chapter is to identify the expectations for the ACS in the first 10 years of implementation and to assess how well the ACS has met the expectations. Additionally, I will review the original scientific basis for the ACS and the relevant science produced since then.

Expectations and Results

Potential Listing of Fish Species and Evolutionarily Significant Units Under the Endangered Species Act

A primary motivation for developing the ACS was the anticipated listing of distinct population segments of various species of Pacific salmon, called evolutionarily significant units (ESUs), and other fish species under the Endangered Species Act (ESA 1973). When the Plan was

Michael J. Furniss

A coho salmon in Bell Creek, in the coastal lakes watershed (Oregon Coast Range) on the Siuslaw National Forest near Florence, Oregon.

developed in 1993, only the Sacramento winter chinook salmon, the shortnose sucker, and the Lost River sucker were listed. Since then, 23 ESUs of Pacific salmon and 3 population segments of bull trout found in the Plan area have been listed. Twenty units of salmon and all bull trout population segments are found on federal lands managed under the Plan (table 9-1). Additionally, the Oregon chub was listed after the Plan was implemented and coho salmon in the Oregon coast is currently a candidate for listing (table 9-1).

The Plan was expected to contribute to the recovery of the ESA-listed fish, particularly the anadromous salmon and trout (that is, fish that spend their early life in freshwater, move to the ocean to mature, and then return to freshwater to reproduce), by increasing the quantity and quality of freshwater habitat (FEMAT 1993). It was not expected to prevent the listing of any species or distinct population segment. The primary reason for this expectation was that the federal land management agencies are responsible only for the habitat they manage; state agencies are responsible for populations on all lands and for the regulation of activities that affect populations and habitats on other ownerships. Factors outside the responsibility of federal land managers contribute to the declines of these populations and will strongly influence their recovery. These

Table 9-1—Evolutionarily significant units (ESUs) of Pacific salmon and trout (*Oncorhynchus* spp.), distinct populations segments (DPSs) of bull trout (*Salvelinus confluentus*), and fish species listed and candidates for listing (*) under the Endangered Species Act that occur in the area covered by the Plan

Species	ESU/DPS	National forests (NF) and Bureau of Land Management (BLM) districts where species occur
Coho salmon	Lower Columbia/southwest Washington	Gifford Pinchot NF, Mount Hood NF
	Oregon coast*	Siuslaw NF, Umpqua NF, Siskiyou NF, Eugene BLM, Coos Bay BLM, Medford BLM, Roseberg BLM, Salem BLM
	Southern Oregon/ northern California	Rogue River-Siskiyou NF, Six Rivers NF, Shasta-Trinity NF, Klamath NF, Mendocino NF, Arcata BLM, Kings Range National Conserva-ion Area (NCA), Redding BLM, Medford BLM, Coos Bay BLM
	Central California coast	Ukiah BLM
Chinook salmon	Puget Sound	Mount Baker-Snoqualmie NF, Olympic NF, Gifford Pinchot NF
	Lower Columbia	Gifford Pinchot NF, Mount Hood NF, Salem BLM
	Upper Columbia	Okanogan NF, Wenatchee NF
	Upper Willamette	Mount Hood NF, Willamette NF, Eugene BLM, Salem BLM
	California coastal	Six Rivers NF, Mendocino NF, Arcata BLM, Kings Range NCA, Ukiah BLM
	Sacramento River winter run	Mendocino BLM
	Central Valley spring run	Shasta-Trinity NF, Mendocino BLM, Redding BLM
	Central Valley winter run	Redding BLM
Chum salmon	Hood Canal summer	Olympic NF
	Columbia River	Salem BLM
Steelhead	Lower Columbia	Gifford Pinchot NF, Mount Hood NF, Salem BLM
	Mid-Columbia	Gifford Pinchot NF, Mount Hood NF, Wenatchee NF
	Upper Columbia	Wenatchee NF, Okanagon NF
	Upper Willamette	Willamette NF, Salem BLM, Eugene BLM
	Northern California	Six Rivers NF, Mendocino BLM, Arcata BLM, Ukiah BLM, Kings Range NCA
	Central California coast	Arcata BLM, Kings Range NCA
	Central Valley, California	Shasta-Trinity NF, Mendocino BLM
Coastal cutthroat trout	Southwest Washington/ Columbia River	Gifford Pinchot NF

Table 9-1—Evolutionarily significant units (ESUs) of Pacific salmon and trout (*Oncorhynchus* spp.), distinct populations segments (DPSs) of bull trout (*Salvelinus confluentus*), and fish species listed and candidates for listing (*) under the Endangered Species Act that occur in the area covered by the Plan (continued)

Species	ESU/DPS	National forests (NF) and Bureau of Land Management (BLM) districts where species occur
Bull trout	Klamath River	Winema NF
	Columbia River	Deschutes NF, Gifford Pinchot NF, Mount Hood NF, Wenatchee NF, Okanongon NF, Willamette NF, Eugene BLM
	Coastal-Puget Sound	Gifford Pinchot NF, Mount Baker-Snoqualmie NF, Olympic NF
Oregon chub		Willamettte NF, Umpqua NF
Lost River sucker		Winema NF
Shortnose sucker		Winema NF

include (National Research Council 1996):

- Degradation and loss of freshwater and estuarine habitats.
- Excessive harvest in commercial and recreational fisheries.
- Migratory impediments, such as dams.
- Loss of genetic integrity from the effects of hatchery practices and introductions.

Ocean productivity also strongly influences population numbers of anadromous salmonids. Conditions in the marine environment in the Plan area are highly variable over time. The oceanic boundary between cool, nutrient-rich northern currents and warm, nutrient-poor southern currents is off the coast of Washington, Oregon, and northern California (Fulton and LaBrasseur 1985) (fig. 9-1). The location of this boundary is influenced by the Pacific Decadal Oscillation (PDO), which is climatically driven and results in an oscillation between positive and negative phases every 20 to 30 years. This oscillation results in alternating regimes of salmon production between the Pacific Northwest and more northerly areas along the Pacific coast of North America (Mantua and others 1997). During periods of high productivity, zooplankton biomass—a critical food for salmonids when they first enter the ocean—is greater in the productive zone than in the less productive region. Early ocean survival of anadromous salmonids and the number of adults returning to freshwater are greater during the positive phases (Mantua and others 1997). The last period of high productivity was from the late 1940s to 1977 (Mantua and others 1997). The Plan area is currently in another positive production phase, but how long the current phase that began in 2001 will last is unknown.

Population numbers of many ESA-listed salmon and trout in the Plan area, and other parts of the Pacific Northwest, have increased since the Plan was implemented. However it is not possible to discern how much the Plan has contributed to this increase. Conditions of freshwater habitats on federal lands have improved moderately under the Plan (see later discussion for more details) but not to an extent that could account for the current increases in the numbers of returning adults. Populations in areas outside of the Plan area have shown similar, and even larger, changes.

The real contribution of freshwater habitats to the persistence and recovery of anadromous salmon and trout in the region covered by the Plan will be measured when the PDO moves into a less productive phase and the persistence of andromous salmon and trout populations will depend to a larger degree on freshwater habitat (Lawson 1993) (fig. 9-2). Improvements in the quantity and quality of

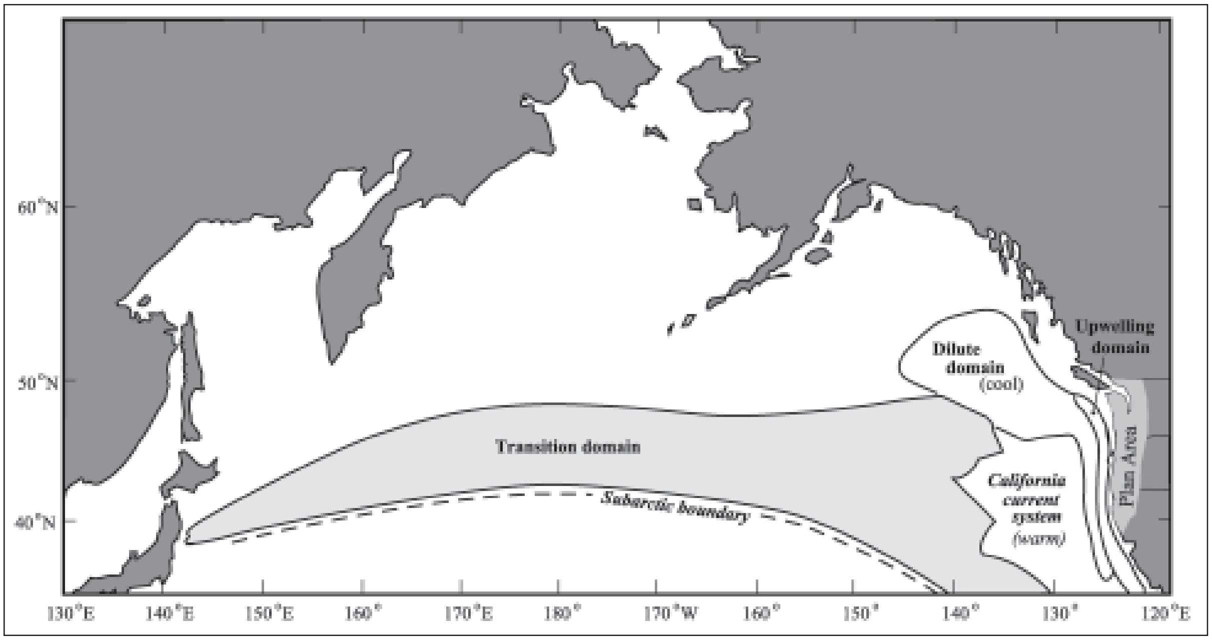


Figure 9-1—Boundaries of eastern north Pacific Ocean currents. Source: Fulton and LaBrasseur 1985.

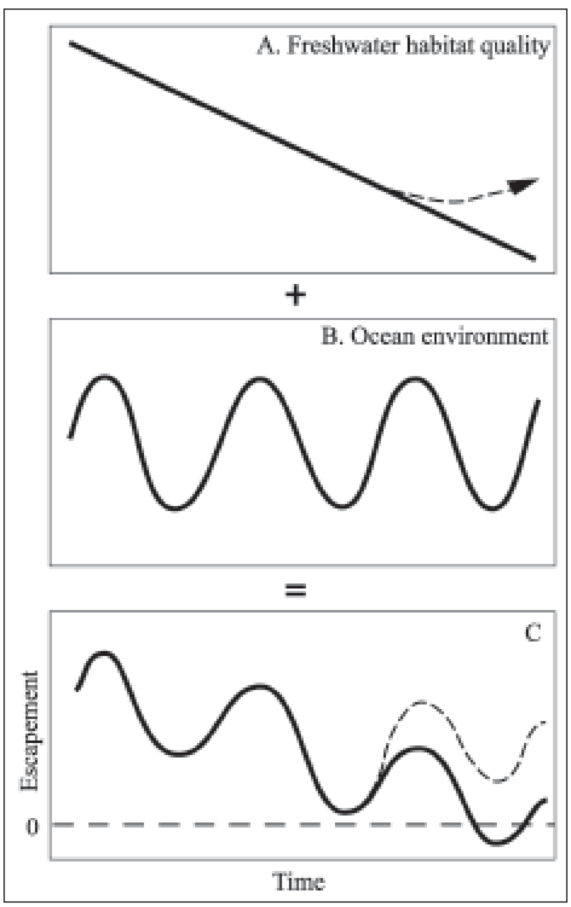


Figure 9-2—Conceptual relation between the quality of freshwater habitat, variable ocean conditions, and the persistence of populations of anadromous salmonids. "A" is the trajectory of habitat quality over time. Dotted line represents possible effects of improvement in habitat quality. "B" is the generalized time series of ocean productivity over time. "C" is the sum of the interaction of A and B. Source: Modified from: Lawson 1993.

freshwater habitat should result in greater numbers of fish entering the ocean, thus increasing the likelihood of persistence of many populations during periods of low productivity.

Changes in Watershed Condition

The ACS was designed to improve the ecological condition of watersheds in the Plan area over an extended time (that is, several years to decades). It is based on preserving key ecological processes and recognizes that periodic disturbances may be required to maintain ecological productivity. As a result, the ACS does not expect that all watersheds will be in good condition at any point in time, nor does it expect that any particular watershed will be in a certain condition through time. If the ACS and the Plan are effective, the proportion of watersheds in better condition is expected to remain the same or increase over time (Reeves and others 2004). However, the ACS does not identify a particular desired or acceptable distribution of watershed condition. It does, however, recognize that significant results from the ACS were not expected for several years or decades because it will take extended time for the condition of watersheds that were extensively degraded from past management activities to improve (FEMAT 1993).

The ACS attempts to improve watershed conditions by preserving key ecological processes.

Large improvements in the condition of individual watersheds or changes in the distribution of conditions were not expected in the short term (10 to 20 years) because this was too short a time for many watersheds to improve, and the impact of restoration efforts would not be extensive enough across the Plan area to result in discernable changes in the distribution of watershed conditions. At best, it was expected that the pattern of degradation would be slowed or halted, and there may be some minor to moderate improvements in watershed condition as a result of the implementation of the ACS.

A monitoring program to determine the effectiveness of the ACS was expected to be developed and implemented within a short time of the record of decision (ROD) (USDA and USDI 1994), but the Aquatic and Riparian Effectiveness Monitoring Program (AREMP) did not begin until 2000. This delay resulted from the difficulty that the relevant agencies (USDA Forest Service [FS], USDI Bureau of Land Management [BLM], the Environmental Protection Agency [EPA], and National Oceanic and Atmospheric Administration [NOAA] Fisheries) had with agreeing on an approach, much less an actual program. Before 2000, two attempts were made to develop an effectiveness monitoring plan that all agencies could support. Both attempts failed because the involved parties could not agree on a common vision for the plan, a common approach to the problem, or methodology. The need for three attempts to develop an effectiveness monitoring plan illustrates the struggle over the ACS because of differences in operating and thinking among the involved agencies. The AREMP was approved by the regional executives in 2000, and pilot testing began that year. Components of AREMP and the rationale for them are described in Reeves and others 2004.

The AREMP attempts to characterize the ecological condition of watersheds by integrating a set of biological and physical indicators, and it tracks the trend in condition of the population of watersheds over time. The condition of watersheds is evaluated with decision-support models by using fuzzy logic (Reeves and others 2004). The relations between the selected parameters and the watershed condition used in these models were based on empirical evidence and the professional judgment of aquatic specialists from the national forests, BLM districts, management and regulatory agencies involved with the Plan, and state fish management agencies. The models were built at the province and subprovince scales to account for ecological variability.

The condition of a watershed was defined as "good" if the physical attributes were adequate to maintain or improve biological integrity, primarily for native and desired fish species (Reeves and others 2004). Also, the systems that were in good condition were expected to be able to recover to desired conditions when disturbed by a natural event or land-management activities. Scores for watershed conditions ranged from 1 to -1: 1 if absolutely true (based on the assumptions in the decision-support model) that the watershed was in good condition, and -1 if absolutely false that it was in good condition. Reeves and others (2004) emphasized the need to recognize that condition of any watershed may vary widely naturally. For that reason, it was recognized that watersheds with little or no human activity were not necessarily in good condition at any point in time.

The focus of AREMP is not on individual watersheds but rather on the statistical distribution of watershed conditions across the Plan area. Two hundred fifty 6^{th}-field watersheds (10,000 to 40,000 acres) were randomly selected from throughout the Plan area to be sampled over a 5-year cycle (Reeves and others 2004). The full range of management from roadless and wilderness to intensive timber harvest and livestock grazing were found in these watersheds.

Pilot testing in AREMP to evaluate sampling protocols and to determine funding and staff requirements occurred in 2000 and 2001. Actual monitoring began in 2002, with about half of the estimated funding needed to fully implement AREMP. Monitoring continued at reduced levels in 2003 and 2004. A total of 55 (of an expected 100) watersheds were sampled in 2002 and 2003 (Gallo and others 2005). No watersheds have been resampled to permit direct estimates of change in watershed condition.

The parameters necessary to estimate watershed condition—in-channel, upslope, and vegetation—were only available for 55 watersheds, and as mentioned above, none of these have been resampled (Gallo and others 2005). Lacking the ability to assess the total changes in watershed conditions in the Plan area, Gallo and others (2005) examined changes associated with riparian vegetation and the amount of roads in the 250 watersheds selected for sampling by AREMP. They calculated partial changes in watershed condition scores based on these parameters for two periods, roughly 1994 and 2003 (fig. 9-3). The distribution of these scores did not change to a statistically significant degree during this time (Gallo and others 2005). This result is not surprising given the relatively short period in which the ACS has been in place and that condition scores only represented a partial change.

The proportion of watersheds (of those that exhibited a change) that had a higher condition in 2003 than in 1994 compared to those with lower scores was greater than expected by chance alone ($P < 0.01$, Wilcoxon signed-rank test [Sokal and Rohlf 1969]). The changes in condition scores for individual watersheds are shown in figure 9-3. The condition scores of about 18 of the 250 remained the same, 161 improved, and 71 decreased between 1994 and 2003 (fig. 9-3). The average changes in scores were relatively small, 0.09 (SD 0.19) for those that increased and 0.14 (SD 0.3) for those that decreased. The decreases in watershed condition scores were not simply related to management activities; the four watersheds that exhibited the largest decline had 30 to 60 percent of the watershed area burned.

The observed changes suggest some progress owing to the ACS. The ecological significance of this progress is not known, however. An understanding of the relation between changes in watershed scores is not established as yet. Also, because there are multiple factors influencing watershed condition, a change in score can occur from a combination of changes in the factors. This is certainly an area that lacks research.

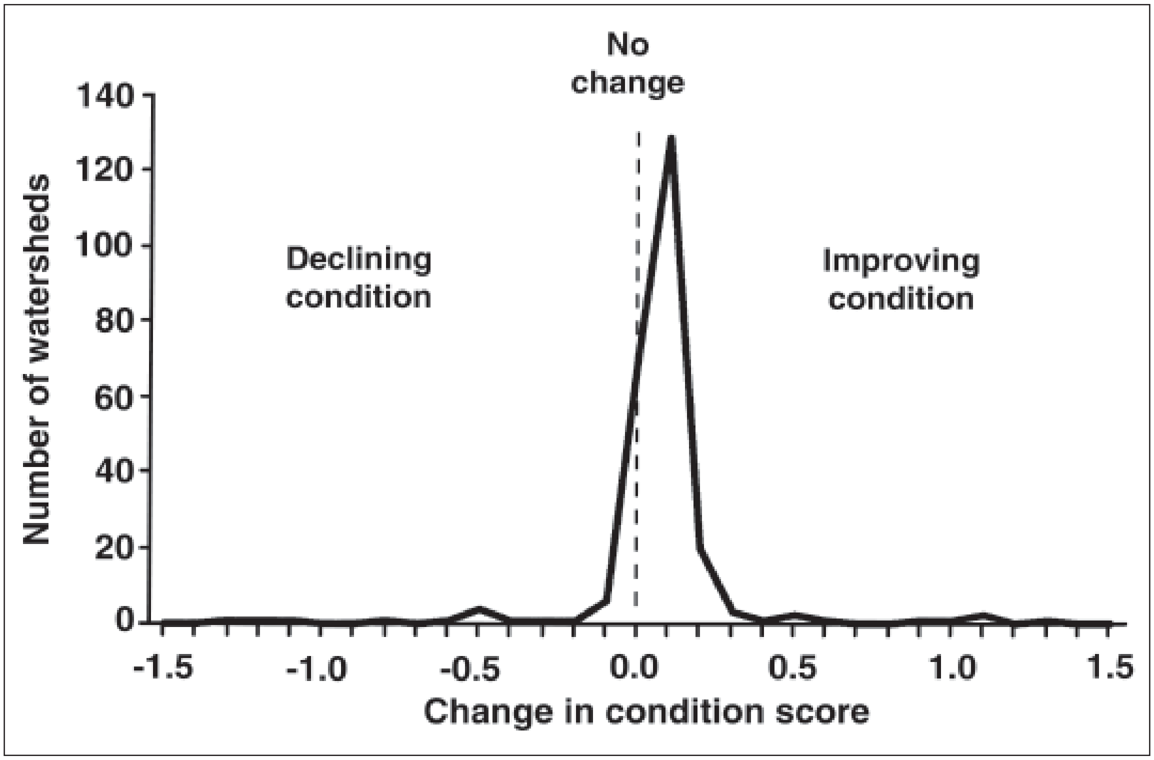


Figure 9-3—Changes in condition scores for 250 watersheds sampled as part of the aquatic and riparian effectiveness monitoring program of the Plan. Source: Gallo and others 2005.

The change in watershed condition scores during the first decade of the Plan was attributable primarily to changes in riparian vegetation and, more specifically, an increase in the number of large trees in riparian areas. The type, size, and distribution of vegetation in riparian and upslope areas influence the condition of aquatic ecosystems (Burnett 2001); generally, the bigger and more numerous the conifers the better the condition of the watershed. Gallo and others (2005) compared the number of trees >20 inches diameter at breast height (d.b.h.) in riparian (defined in the ACS as 150 feet on both sides of the stream on the west side of the Plan area and 90 feet on the east side) and upslope areas in the 250 watersheds in 1996 shortly after the Plan was implemented and in 2002. They used the geographic information system (GIS) layers developed by the Interagency Vegetation Mapping Project (IVMP) for Oregon and Washington and CalVeg for California, which were used to assess changes in late-successional and old-growth habitat (Moeur and others 2005). The number of large trees increased an estimated 2 to 4 percent during this time, most likely the result of tree growth into the >20-inch d.b.h. category (Gallo and others 2005). Concurrently, the amount of riparian area subjected to clearcutting on federal lands in Oregon and Washington in the Plan area was one-seventh the level of harvest in 1988-91 and even less compared to earlier periods (Gallo and others 2005). Projections of tree size on federally managed lands in the central and northern Oregon Coast range suggest that the number of large trees will continue to increase by 15 to 20 percent over the next 100 years under the current policy (Burnett and others, in press; Spies and others, in press).

Roads, permanent and temporary, can significantly affect aquatic ecosystems. They can result in increased rates of erosion (Furniss and others 1991, Potondy and others 1991), which, in turn, may affect populations of fish and other aquatic organisms (Quigley and Arbelbide 1997, Young and others 1991) and their habitats (Buffington and others 2002, Megahan and Kidd 1972). They can also form barriers to movements and can reduce interactions within and among populations of fish, amphibians, and other aquatic organisms (Trombulak and Frissell 1999).

The condition scores of watersheds as influenced by roads generally did not change significantly since the Plan was implemented (Gallo and others 2005). Three of the watersheds that had the largest increase in condition scores had the most extensive road decommissioning efforts (Gallo and others 2005). It is likely in the other cases that the amounts of road removed from any given watershed may have been relatively small and insufficient to change the watershed condition. There were 3,324 miles of road (3.6 percent of the total road mileage) decommissioned from 1995 to 2002 on FS and BLM lands (Baker and others, in press). An estimated 354 miles of new roads were constructed during the same time (Baker and others, in press). The effect of roads on aquatic ecosystems is also a function of road location; valley bottom roads affect aquatic ecosystems more than those on ridgetops (Wemple and others 2001). The provincial and subprovincial models that evaluate watershed condition differed widely in how they considered road location; some consider location, whereas others only consider the density of roads. Modification of those that currently do not consider road location may increase their sensitivity to restoration activities.

Several miles of roads have been “improved”—that is, actions were taken to reduce sediment delivery and improve stability or to allow more natural functioning of streams and flood plains, which includes improvements in drainage, stabilization, and relocation (Baker and others, in press). However, the watershed condition models currently do not take this into account because road improvement data are currently not available in the federal agencies’ corporate databases.

Assessment of the ecological condition of an individual watershed was done on the basis of the entire landscape, which resulted, in many instances, in considering conditions on nonfederal lands. In many of the watersheds sampled by AREMP, there were a number of different owners, each with objectives and practices that differed from those of the Plan. Watersheds with more nonfederal ownership had the lowest changes in watershed condition scores (Gallo and others 2005). This influences the potential amount of change that can be expected in some watersheds and could be considered in future assessments of the effectiveness of the ACS.

One clear success of the ACS is a change in the general expectation of trends in aquatic conditions across the Plan area. There is general recognition that aquatic conditions deteriorated during the pre-Plan periods of intensive federal timber harvest and road building, and these declines were predicted to continue under many of the forest plans that the Plan amended. Several forest plans that were to be implemented before the Plan acknowledged that aquatic habitat would decline (for example, the Siuslaw National Forest [NF]) or have a high probability of declining (Umpqua NF, Siskiyou NF). Many of the activities that could have had negative effects on aquatic ecosystems, however, have decreased under the Plan. As cited earlier, the amount of timber harvest in riparian areas decreased substantially (Gallo and others 2005). Implementing the ACS appears also to have influenced the rate at which roads were built in the Plan area. The amount of roads decommissioned was 10 times the amount built between 1995 and 2002, the reverse of the trend before the Plan (Baker and others, in press). The ACS and the Plan appear to have prevented further degradation of watersheds that was likely under previous forest plans.

Riparian Reserves

The riparian reserve network established by the ACS encompasses an estimated 2.6 million acres (Baker and others, in press) and was one of the major changes from previous forest plans. Before the ACS, the riparian ecosystem was generally defined as 100 feet on either side of fish-bearing streams and some areas with high landslide risk. The riparian reserve network of the ACS was based on an “ecological functional” approach that identified zones of influence rather than set distances and included the entire stream network, not just fish-bearing streams. Consequently,


Example of how riparian habitat extends from the edge of a stream.

the riparian zone along streams was expanded to the height of two site-potential trees (or 300 feet) along fish-bearing streams and one tree height (or 150 feet) along permanently flowing and intermittent non-fish-bearing streams (USDA and USDI 1994). The latter undoubtedly contributed the greatest to the increased amount of area considered as the riparian reserve. More than 800 of the more than 1,100 organisms considered in FEMAT (1993) were found to be associated with the riparian reserve network. It was also suggested in FEMAT (1993) that the width of the riparian reserve on each side of headwater streams be equal to one-half the height of a site-potential tree, but it was changed to a full tree height in the ROD (USDA and USDI 1994) to increase the likelihood of persistence of habitat for aquatic and riparian-dependent organisms.

The initial riparian reserve network was expected to be interim, and activities within them were very restricted until a watershed analysis was completed. It appears, however, that the interim boundaries of the riparian reserves remained intact in the vast majority of watersheds (Baker and others, in press). The primary reasons offered for the relatively low harvest in the riparian reserve were that it was difficult to justify changing the interim boundaries or that there was no compelling justification for changing the interim boundaries. (It should be noted that harvest from the riparian reserve was not part of the estimates of potential timber harvest.) Baker and others (in press) found that agency personnel thought that “burden of proof [for changing interim boundaries] was too high.” No explicit criteria for changing the boundaries were offered by the Forest Ecosystem Management Assessment Team (FEMAT 1993) or the ROD (USDA and USDI 1994), but tools are available now that can help identify the more ecologically important parts of the riparian and stream network from an aquatic perspective (such as Benda and others, n.d.). Because watershed analysis is an interdisciplinary endeavor, however, changes in the riparian reserve boundaries need to consider non-aquatic factors such as terrestrial and social concerns. Only a few watershed analyses considered these factors (such as Cissel and others 1998). The effect of the extent of the riparian reserves is probably most likely in the steeper more highly dissected landscapes, where the riparian reserves network is most extensive (FEMAT 1993).

Timber production, primarily in precommercial thinning, has occurred on an estimated 48,000 acres (1.8 percent of the estimated total area) of the riparian reserve (table 9-2). The volume of timber harvested is not known because agencies do not track it. Timber harvest was expected to occur in riparian reserves, but no level was specified by FEMAT (1993) or the ROD (USDA and USDI 1994). Harvest from the riparian reserve was not part of the estimated potential sale quantity of the Plan. Agency personnel thought that one of the primary reasons for the limited timber harvest in the riparian reserve was the difficulty in changing boundaries and in determining that there would be no adverse affects from the activities (Baker and others, in press).

Watershed Restoration

Watershed restoration efforts were expected to be a catalyst for initiating ecological recovery (FEMAT 1993). It was expected that restoration efforts would be comprehensive, addressing both protection of existing functioning aspects of a watershed and restoration of degraded or compromised aspects. It was recognized that it may not be possible for restoration efforts to restore every watershed or that some

Table 9-2—Estimated area of riparian reserve in which silvicultural activities have occurred during the first 10 years of the Plan

Administrative unit	Period	Treatment: Precommerical thin	Treatment: Regeneration harvest	Total
		Acres		
USDA Forest Service				
Region 6				
Mount Baker-Snoqualamie	1994-2000	1,100	0	1,100
Okanogan-Wenatchee	1994-2000	875	300	1,175
Gifford-Pinchot	1994-2004	600	0	600
Olympic	1994-2004	1,100	1,100	2,200
Mount Hood	1998-2004			1,200[a]
Deschutes	1997-2004	700	0	700
Willamette	1994-2004	6,600	125	6,725
Siuslaw	1994-2004	1,285	12,570	13,855
Umpqua	1994-2004	2,200	300	2,500
Siskiyou-Rogue River	2000-2004	1,902	0	1,902
Fremont-Winema	2003	0	0	400[b]
Estimated total		16,362	14,395	32,357
Region 5				
Klamath	1994-2004	4,598	781	5,379
Shasta-Trinity	1994-2004	1,701	515	2,216
Six Rivers	1994-2004	3,288	516	3,804
Mendocino	1994-2004	0	0	0
Estimated total		9,587	1,812	11,399
Bureau of Land Management				
Oregon-Washington				
Salem	1995-2003			797[b]
Coos Bay	1995-2003			1,326[b]
Eugene	1995-2003			520[b]
Roseburg	1995-2003			827[b]
Medford	1995-2003			663[b]
Estimated total				4,133
California				
Arcata	1995-2004	84	0	84
Ukiah	1995-2004	0	0	0
Estimated total		84		84
Estimated total				47,973

[a] Estimate was of 100 to 200 acres per year with no breakdown of treatment type.
[b] No breakdown of treatment type provided.

would only have limited success because of the extensive level of degradation. The impact of restoration efforts was not expected to be large or to be immediately visible. At the watershed scale, it may take an extended time to observe the effect of the restoration effort. The aggregate effect of watershed restoration effort, particularly those done during the initial phases of the ACS, may not be observable at the regional scale. Although it may appear that relatively large amounts of area have been restored, the reality is that this represents a small part of the total area that is degraded.

Paul Burns

A restoration project on Fiddle Creek (Siuslaw National Forest) where a portable yarder was used to pull logs into the creek from surrounding mature Douglas-fir stands to enhance spawning and rearing habitat for coho salmon.

It is not possible to accurately assess the regional effect of the numerous restoration efforts undertaken as part of the ACS. Gallo and others (2005) highlighted several watershed restoration efforts that were successful, but their impact cannot be discerned at the regional scale. The length of streams restored or made assessable to fish is also a relatively small fraction of the totals. However, the watersheds that had the largest improvement in condition scores were three that had relatively extensive road restoration programs (Gallo and others 2005). Similarly, Baker and others (in press) reported that almost 69,000 acres of riparian reserve were restored, primarily in Washington and Oregon, between 1998 and 2003. The total amount of area in riparian reserve in this area is not known, but the 69,000 acres represents a relatively small part (estimated at about 2.6 percent) of total area occupied by the riparian reserve. It is expected that as time passes, the effect of these restoration efforts that have been implemented already and those that may occur in the future will be more discernable.

Key Watersheds

Key watersheds (1) are intended to serve as refugia for aquatic organisms, particularly in the short term for at-risk fish populations; (2) have the greatest potential for restoration; or (3) provide sources of high-quality water (USDA and USDI 1994). Tier 1 key watersheds serve one of the first two purposes. These include 141 watersheds covering 8.1 million acres. Tier 2 key watersheds provide sources of high-quality water and include 23 watersheds covering about 1 million acres. Key watersheds were aligned with late-successional reserves as closely as possible to maximize ecological efficiency (USDA and USDI 1994) and to minimize the amount of area in which timber harvest activities were restricted.

A primary objective for the Tier 1 key watersheds was to aid in the recovery of ESA-listed fish, particularly in the short term (FEMAT 1993). Refugia that are areas of high-quality habitat and contain remnant populations are a cornerstone of conservation strategies. Past attempts to recover fish populations were generally unsuccessful because the focus was on fragmented areas of good habitat in stream reaches and not on a watershed perspective (Moyle and Sato 1991, Naiman and others 1992, Williams and others 1989). Tier 1 key watersheds currently in good condition were assumed to serve as anchors for potential recovery of depressed populations. Tier 1 key watersheds that had degraded conditions were judged to have the greatest potential for restoration and therefore become future sources of good habitat.

Key watersheds had greater increases in condition scores than did non-key watersheds (Gallo and others 2005). More than 70 percent of the key watersheds improved, whereas less than 50 percent of the non-key watersheds improved. The primary reason was that more than twice as many miles of roads were decommissioned in key watersheds compared to non-key watersheds. This result suggests that land management agencies appear to have treated key watersheds as priority areas for restoration, as stated in the ROD (USDA and USDI 1994).

Key watersheds were originally selected based on the professional judgment of fish biologists from the national forests and BLM districts covered by the Plan. No formal evaluation of the potential effectiveness of the network was conducted when the Plan was developed or since it was implemented. Fish populations in need of attention are clearly identified now, and it would be useful to see if the current system is beneficial to those fish in terms of the overall distribution as well as the suitability of individual watersheds.

New techniques are now available to aid in this assessment. For example, Burnett and others (2003) have developed a process to identify the potential of a watershed or stream reach to provide habitat for coho salmon and steelhead based on topographic features. In an analysis of a portion of the northern Oregon Coast Range, areas with the highest potential to provide habitat for coho salmon, an ESA candidate species, were primarily on private lands and for steelhead, which is not a listed species, on public lands. Analysis of Oregon State, BLM, and FS Pacific Northwest Region (R6) Forest Service Lands in the Oregon Coast Range (Peets and Doelker 2005) found that about 10 percent (155 miles) of the area with the best potential to provide habitat for coho salmon was on federally managed lands. A relatively small proportion of this habitat is found in key watersheds. Similar analyses in other areas could help determine the current effectiveness of the key watersheds.

Watershed Analyses

Watershed analysis was intended to provide the context for management activities in a particular watershed. It was to serve as the basis for developing project-specific proposals and determining restoration needs. It was envisioned in the ROD (USDA and USDI 1994) as analysis to involve individuals from the appropriate disciplines but not a decision-making process. The management agencies were expected to complete a watershed analysis before activities (except minor ones) were started in key watersheds and riparian reserves (USDA and USDI 1994b). The version of watershed analysis advocated in the Plan differed from the versions of watershed analyses that were used at the time (such as the Washington Forest Practices Board 1993) in that it involved disciplines and issues other than aquatic ones. Since the ROD (USDA and USDI 1994), several publications have examined the watershed analysis process and framework (Montgomery and others 1995, Reid 1998), but these analyses have been primarily from an aquatic perspective. A more comprehensive review and evaluation of watershed analyses could help improve processes and likely reduce costs while increasing the usefulness of the product.

Baker and others (in press) estimated that 89 percent of the watersheds (of a total of 550 watersheds) in the Plan area had completed watershed analyses by 2003 and that some unknown proportion of them had been revised at least once. This percentage seems high, given budget and personnel constraints that the land management agencies have faced. No formal assessment of watershed analyses has been done, but their quality and effectiveness likely differ widely. There is also the opportunity to reexamine the watershed analyses process to see if it can be conducted more efficiently and include not just a focus on the watershed of interest and what happens there but the context of the watershed in the basin. The latter is particularly relevant for the Plan to be implemented at a landscape scale.

Relevant New Science Information

Landscapes and Dynamic Ecosystems

The ACS was based on the best science available at the time. Much scientific literature on aquatic ecosystems, on the effects of human activities on them, and on conservation strategies for fish and other aquatic and riparian organisms has been produced since the Plan was implemented in 1994. Key science findings on the ecosystem and landscape dynamics and the historical range of variation (HRV) and on the ecological role of headwater streams are summarized here. These topics relate to ACS components and are particularly relevant to assessing the validity of the ACS components and other parts of the Plan and for considering future modifications. Not all of the relevant scientific literature is summarized or reviewed here. Documents that provide excellent reviews and synthesis on these and other relevant topics include Spence and others (1996), Naiman and Bilby (1998), National Research Council (1996), Gresswell (1999), and Everest and Reeves (in press.).

The ACS combined ecosystem and landscape perspectives to forge a management strategy that could be applied over broad heterogeneous areas. Before the ACS was developed, much of the management and research focus for fish ecology and conservation was on relatively small spatial scales, such as habitat units (Bisson and others 1982, Nickelson and others 1992) and reaches (Murphy and Koski 1989). At these scales, the needs of individual fish or communities are the primary interest. Williams and others (1989) found that no fish species listed under the ESA was ever recovered after listing and attributed this failure to the general focus of recovery efforts on habitat attributes rather than on restoring and conserving ecosystems. Thus, the developers of the ACS believed that shifting the focus to larger scales was necessary to aid in the recovery of freshwater habitats of listed and declining populations of anadromous salmon and trout and other fish in the range of the northern spotted owl. Since the ROD was approved (USDA and USDI 1994), a variety of sources, including interested citizens, interest groups, scientific review and evaluation groups (such as the Independent Multidisciplinary Scientific Team 1999, National Research Council 1996), regulatory agencies, and policy- and decisionmakers have called for developing policies and practices to manage the freshwater habitats of at-risk fish at ecosystem and landscape scales.

Understanding the differences and relation between scale and ecological organization is critical to implementing and evaluating the ACS. Allen and Hoekstra (1992) proposed a framework that emphasizes the role of the observer in choosing a scale of observation and deciding how to conceptually organize the parts and processes. By **scale**, they mean spatial or temporal extent. In contrast, **organization** is a subjective or definitional construct that invokes implicit, user-defined criteria. Ecological organization, such as ecosystem, landscape, or population, has meaning without any reference to a particular scale. For real-world management issues, both scale and organization should be made explicit. The intersection of the two creates a clear conceptual boundary that allows discourse and management to proceed.

Ecosystems and landscapes are levels of organization that are especially important within the ACS. Of the two, landscapes are the most tangible in that spatial proximity is the organizing principle (Allen and Hoekstra 1992), and the components of the landscape (such as forest stands, streams, clearings, roads, and so on) are readily apparent to human observers. From an aquatic perspective, the landscape of interest can be quite large and include multiple watersheds (Reeves and others 2002, 2004) but spatial patterns (that is, landscape attributes) can also be important at smaller scales. In contrast to landscapes, ecosystems are organized around the interaction between physical and biological components. The processes and material flows that are the substance of the ecosystem organization may be difficult to observe. Reeves and others (2002, 2004) used the directional flow of water to define aquatic ecosystems, and bounded their spatial extent by using watersheds, defined by FEMAT (1993) as subbasins of 20 to 200 square miles.

In conventional terms, ecosystem management often refers to managing large geographic areas, which has contributed to the confusion between ecosystems and scale. Lugo and others (1999) reiterated the major paradigms of ecosystem management, including:

- Ecosystems are not steady state but are constantly changing through time.
- Ecosystems should be managed from the perspective of resilience, as opposed to stability.
- Disturbance is an integral part of any ecosystem and is required to maintain ecosystems.

Clearly, these principles are not tied to a particular scale and would apply equally well to a single watershed and to a region.

Ecologists and managers recognize the dynamic nature of terrestrial ecosystems and how the associated biota and physical characteristics change through time. They are also aware that the range of conditions an ecosystem experiences is determined to a large extent by the disturbance it experiences (such as wildfire, hurricane, and timber harvest and associated activities). Natural disturbances can increase biological diversity, be crucial for the persistence of some organisms and the habitat that support them, and express and maintain key ecological processes (Turner and others 1994). Disturbances invariably involve a disruption in existing connections among ecosystem components, which leads to the release of nutrients and other materials and the potential for reorganization (Holling 1992). Resilience is the ability of an ecosystem to recover after a disturbance (Lugo and others 1999). An ecosystem demonstrates resilience after a disturbance when the environmental conditions after the disturbance are within the range of conditions that the system exhibited before the disturbance. Reduced resilience may result in both the extirpation of some species and increases in species favored by available habitats (Hansen and Urban 1992, Harrison and Quinn 1989, Levin 1974).

Given the role of disturbance in ecosystem dynamics, it is reasonable to expect ecosystems to be most resilient to the types of disturbance under which an ecosystem developed. Thus, one approach to minimizing management impacts is to make the combination of management actions and natural disturbance resemble the natural disturbance regime as closely as possible (Lindenmayer and Franklin 2002). Factors considered in developing ecosystem management plans and policies include the frequency, magnitude (Hobbs and Huenneke 1992, White and Pickett 1985), and legacy (that is, the conditions and materials that exist immediately following the disturbance) (Lindenmayer and Franklin 2002, Reeves and others 1995) of disturbance regimes in managed ecosystems. The effects of land management on the ecosystem depend on how closely the management disturbance regime resembles the natural disturbance regime with regard to these factors. Everest and Reeves (in press) reported they found little evidence or studies in the peer-reviewed literature of fish populations or habitat responding positively to or remaining unchanged as a result of intensive land management activities.

Landscape management strives to maintain a variety of ecological states in some desired spatial and temporal distribution. Management at that scale addresses the dynamics of individual ecosystems, the external factors that influence the ecosystems that compose the landscape, and the dynamics of the aggregate of ecosystems (Concannon and others 1999). To do this, landscape management could consider developing a variety of conditions or states in individual ecosystems within the landscape and the pattern resulting from the range of ecological conditions that are present (Gosz and others 1999). The specific features of the ecological states and their temporal and spatial distribution will vary with the objectives for a given landscape.

Scientists and managers have worked in concert to try to develop tools and techniques to facilitate landscape management. One such approach relies on HRV, which is conditions that a level of organization experiences naturally over an extended time, from several decades to centuries. The term is often used for individual components of an ecosystem, such as the number of pieces of large wood or number of pools, or for ecological states. The usual manner for establishing the HRV for a component of interest is to

Pete Bisson

Streams with the greatest diversity of juvenile salmonids can be in midsuccessional forests.

measure the parameter in pristine systems (systems with little or no history of effects from human activities). The HRV is represented by the distribution of these values. This range is well established for terrestrial systems (early-, mid-, and late-successional) (for example, Wimberly and others 2000), but it is not incorporated into aquatic ecology.

Spatial scale is an important, but not well recognized, element of the HRV. The HRV is generally inversely related to spatial scale (Wimberly and others 2000) because it represents the range of average condition for the area. The smaller the spatial scale, the larger is the HRV and, conversely, the larger the scale, the smaller the HRV. Hierarchy theory provides the rationale for this relation and is an appropriate framework for considering ecosystem issues at and between different spatial scales (Overton 1977). Each level in the hierarchy of an ecosystem has unique properties and behaviors that are expressed over time. The properties of lower levels of organization are "averaged, filtered, and smoothed" as they are aggregated at higher levels of organization (O'Neill and others 1986). Consequently, the range and variability in the properties and conditions of the system are relatively wide at lower levels of organization compared to higher levels (Wimberly and others 2000). A recent paper on the concept of HRV (Landres and others 1999), and another estimating HRVs (Keane and others 2002) did not consider the effect of spatial scales.

Wimberly and others (2000) illustrated the HRV of successional vegetative stages in the Oregon Coast Range at multiple spatial scales. They estimated (based on a model of fire frequency and intensity and vegetation response over 3,000 years) that, at the scale of a late-successional reserve (100,000 acres), the range in the amount of old growth was from 0 to 100 percent. For an area roughly the size of a national forest (750,000 acres), the HRV for old-growth was from about 10 to 75 percent. The HRV for the Coast Range (5,600,000 acres) was 30 to 55 percent. The large, infrequent disturbance events generally affect relatively small portions of the landscape at any one time. Thus, having the entire area affected by a disturbance event at the same time is highly unlikely. The asynchronous nature of the disturbance events results in a series of patches of vegetation of different ages. This narrows the HRV because of the reduced likelihood of finding the entire area either with no or all old-growth at any particular time. The HRV is further reduced at larger spatial scales because disturbance events are even more desynchronized. Consequently, the range and variability in the properties and conditions of the system are relatively wide at lower levels of organization compared to higher levels (Wimberly and others 2000).

Spatial scale and implementation problems—
The developers of the Aquatic Conservation Strategy (FEMAT 1993) and the ROD (USDA and USDI 1994) did not fully recognize the implications of shifts to the landscape scale of the Plan and the ACS and its objectives, which has led to much confusion with the ACS objectives. The land management and regulatory agencies initially attempted to meet all of the ACS objectives for any action, which led to many problems and was the impetus for the final environmental impact statement (FSEIS) that clarified the intent of the ACS (USDA and USDI 2003). The objectives provide a framework for managing aquatic ecosystems at multiple spatial scales, but they became a checklist to evaluate the acceptability of any proposed action at the site scale. The objectives were not intended to be a hard set of criteria that could be applied equally at

each spatial scale of concern. This application was technically impossible because the objectives include a range of spatial scales, and the relation among scales was not considered. For example, objectives 1, 2, and 9 (listed on page 1) deal with landscape and regional objectives. The others deal with ecosystems. Determining consistency with the ACS at the site or small watershed scale is not as simple as assuming that all sites or small watersheds need to be in "good" condition at all times and that any actions that "degrade" a site or small watershed violates the ACS objectives. Conditions at the small scale range widely over time. The overriding objective is to have a mix of conditions at the broader scale, which requires that individual sites each exhibit a range of conditions over time.

Consistency at the small scale (site or subwatershed) is determined by the range of variability established at the larger scales (watershed or basin). The range of variability at the larger scales is the frequency distribution of conditions at the smaller scale that support acceptable amounts of habitat for populations of fish and other aquatic organisms. Watershed analysis was expected to establish the range of variability at the different scales, which was to be used to determine if proposed actions were consistent with the ACS. The focus of watershed analyses, however, has been primarily on the watershed; they fail to provide the context of the watershed in the larger landscape.

The recent supplemental FSEIS that clarifies the original intent of the ACS (USDA and USDI 2003) discusses the importance of considering multiple scales. Dealing with this issue is important if the ACS is to succeed.

Dynamics and aquatic ecosystems—

The perspective that aquatic systems are dynamic, particularly at the ecosystem and landscape scales, was not widely recognized, and no time was left to work out the implications when the ACS was developed. Before it was developed, a small number of researchers recognized that biotic (Resh and others 1988) and physical (Swanson and others 1988) components of aquatic systems, particularly at the smaller spatial scales, were influenced by relatively infrequent events, such as floods. One reason for the absence of the recognition of dynamics of aquatic ecosystems is that the major paradigms that shape our thinking about aquatic systems, such as the River Continuum Concept (Vannote and others 1980), do not consider time or its influence. Similarly, classification schemes such as that of Rosgen (1994) identify a single set of conditions for a given stream or reach type; how these conditions may vary over time is not considered. The physical and biological relations were assumed to be fixed in time and to be unchanging. From this perspective, watershed processes were assumed to be continuous and predictable, implying that the biophysical changes along the riverine network were easily predictable and modeled (for example, Newbold and others 1982, Vannote and others 1980).

Frissell and others (1986) described the hierarchical organization of aquatic ecosystems and identified a temporal component associated with each spatial scale; the finer the scale, the shorter the response period. However, they did not consider how features of a given level in the hierarchy respond over time. A more recent examination of the hierarchical organization of streams by Fausch and others (2002) also recognized that time is a critical factor to consider when examining aquatic ecosystems. They did not integrate time into their description of stream systems, however. The failure to incorporate time into consideration of aquatic systems, especially at higher levels of organization, has led to an implied expectation that stream ecosystems experience a limited, if not a single, set of conditions and that this condition is relatively stable through time.

The foundation for the ACS focus on ecological processes and dynamics came from Naiman and others (1992). They hypothesized that different parts of a watershed (headwaters, middle portion, and lower portion) had different disturbance regimes, based on the frequency and magnitude of disturbance. They also believed that the landscape would have watersheds with a range of conditions because of the asynchronous nature of large and infrequent disturbance events, such as wildfire and flooding. More recent studies have proposed that stream systems are complex networks with branched shapes rather than linear systems, which provides a better understanding of the ecological processes that link riparian and aquatic ecosystems (Benda and others 2004, Fisher 1997). This perspective implies that aquatic ecosystems are not steady state; rather, streams are invariably dynamic, and their conditions vary in space and time because of periodic events such as wildfire and large storms and subsequent floods, hillslope failures, landslides, and debris flows. The signatures of these events are most visible at tributary junctions, which also are sites of high biological diversity (Benda and others 2004).

Since the Plan was implemented, several studies examined the dynamics of aquatic ecosystems in space and time. Reeves and others (1995) described the range of conditions of watershed in the Tyee sandstones of the central Oregon coast in response to wildfire. They found a range of conditions from less productive to more productive. The most complex habitat and biologically diverse fish assemblage was found in a stream that was about 160 to 180 years from the last major wildfire disturbance. Simplified habitat conditions and less diverse fish assemblages were found in streams that were more recently disturbed (80 to 100 years) and that had not been disturbed for a longer period (300+ years). This pattern appears to have resulted from the change in amounts of wood and sediment over time. Immediately after a wildfire, channels are filled with sediments and, as result, much of the wood is buried. The amount of sediment decreases over time because it is eroded and exported from the system faster than it is being delivered to the channel from hillslopes stabilized by forest recovery. Habitat conditions improve as the amount of sediment declines and wood increases either from recruitment or excavation. After extended times, however, sediment declines to amounts that do not support development of pools.

Headwater streams in the same region studied by Reeves and others (1995) exhibited a different pattern of variation in conditions over time (May and Gresswell 2004). Channels that had not been disturbed for several decades were filled with gravel and wood. Recently disturbed channels were devoid of sediment and wood and were scoured to bedrock. Benda and Dunne (1997a, 1997b) and Benda and others (1998) described a similar distribution of in-channel sediment conditions in watersheds over time. Benda and others (2003b) examined the effects of landslides after wildfires on aquatic ecosystems in the Boise River, Idaho. The landslides significantly affected the channel, creating complex channels and delivering large amounts of wood to the channel. As was observed in the Oregon Coast Range (Reeves and others 1995), channel conditions are expected to vary widely over time. See box on next page for further discussion on the variation among watersheds in the response to large disturbance events.

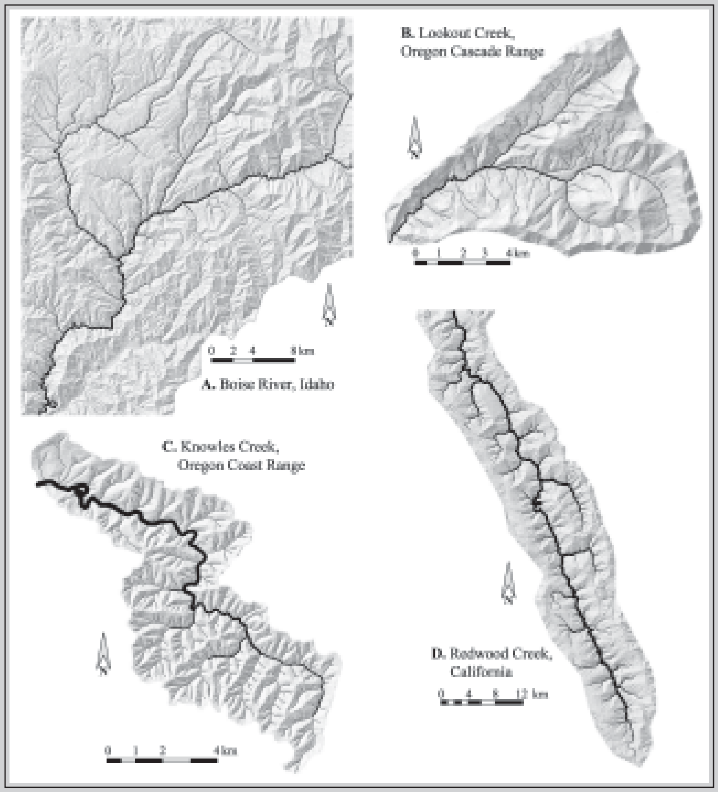
Several factors influenced the responses of these studies. The physical legacy of the disturbances was important; wood in headwater channels accumulated gravel and began the refilling process. Wood and sediment delivered to fish-bearing streams from headwater channels facilitated development of conditions favorable to fish over time. Refugia can be areas that afford protection to individuals during the disturbance event and in the affected area or in nearby areas that are not affected and provide sources of individuals to reestablish populations in affected areas (Roghair and others 2002, Sedell and others 1990). The life history (Dolloff and others 1994) and habitat requirements (Reeves and others 1993, 2002) can also influence the immediate and long-term responses of a population to disturbance events.

Implications—

The dynamic view of aquatic ecosystems and landscapes just described is at odds with the experience and perspectives of some in the research, management, and regulatory agencies and the public. Montgomery and others (2003) questioned the role that dynamics play under natural conditions. They contend that the role of disturbances such as debris flows in old-growth forests is limited. They believe that models of disturbance ecology for salmonids, such as that presented by Reeves and others (1995), need to recognize differences in the disturbance dynamics of old-growth and industrial forests to "provide credible avenues

Variation in Susceptibility to and Response of Watersheds in the Northwest Forest Plan Area to Natural Disturbances

The recognition that dynamic processes, such as periodic large disturbances, have strong impacts on aquatic ecosystems represents a relatively new perspective (for example, Naiman and others 1992, Resh and others 1988). Moderate to large-scale fluctuations in the movement and storage of sediment and wood during these events can create habitats and features that have long-term implications for system productivity (Benda and others 2003b). There is wide variation in the response of aquatic ecosystems to given disturbance events depending on the frequency and magnitude of the disturbance event and a watershed's local topography, channel type (Montgomery and Buffington 1993), shape and configuration of the stream network (Benda and others 2004), and soil and rock type. The four watersheds shown here illustrate some of this variation. The North Fork of the Boise River (A) is outside the Plan area but is representative of parts of the dryer portions of the Plan area. In these steeper systems, periodic disturbances are relatively frequent because of wildfires, but the disturbances have moderate impacts on the channel, and the system is relatively resilient. Postfire sedimentation can lead to large-scale channel changes in small streams and local changes in large channels at tributary confluences (Benda and others 2003a).



Figures from L.E. Benda. 2005. Geomorphologist, Earth Systems Institute, Mount Shasta, CA.

Lookout Creek (B) is on the west side of the Cascade Mountains. It is in an area of hard rock and has a relatively limited stream network. Additionally, the channel gradient is relatively steep. Wildfires and floods, the primary natural disturbances, are relatively infrequent but large. The channel is generally resilient to disturbances, except at some lower gradients spots within the network. The range of conditions observed within the channel is relatively limited.

Knowles Creek (C) is in the soft rock Tyee sandstones of the central Oregon coast, similar to the streams studies by Reeves and others (1995). The primary natural disturbances are infrequent, but large, floods and wildfires. The watershed is characterized by relatively steep tributaries and a lower gradient main channel. The latter results in the deposition of large amounts of wood and sediment in the channel, which experiences a wide range of conditions over time as a result of disturbances events.

Redwood Creek (D) is in northern California. The basin is long and narrow and has a large natural sediment load. The upper portion of the basin is relatively narrow so material moves through it relatively quickly; as a result, inchannel conditions are relatively stable. The lower end is lower gradient and, as a result, is a depositional area. Consequently, there can be a wide variation in habitat conditions over time.

for determining risk associated with land management in steep forested terrain" (Montgomery and others 2003). They believe that "management recommendations based on evolutionary interpretations that are themselves based on a disturbance model primarily applicable to industrial forests may prove misleading" (Montgomery and others 2003).

Clearly, obstacles remain in the path toward a fully implemented ACS that is consistent with the vision articulated in FEMAT (1993) and the ROD (USDA and USDI 1994). Experience has shown that the ACS accommodates a management model that is an alternative to site-specific standards and guidelines. Reeves and others (1995, 1998, 2002) presented an example for the Oregon Coast Range. Another example was for the central Oregon Cascade Mountains (Cissel and others 1998). Progress could be facilitated by attention to several pressing issues.

Focusing policies for and management of aquatic ecosystems at the landscape scale presents challenges to policymakers, managers, and regulators (Reeves and others 2002). A fuller exposition of the HRV would provide a richer understanding of how the conditions of aquatic ecosystems vary through time at all spatial scales and the ecological, social, and economic implications of this variation. Currently, the historical range of the conditions of aquatic ecosystems is assumed to be small and, generally, to be good for habitat. Many managers, regulators, and interested citizens expect aquatic conditions to be relatively constant through time and to be good in all systems at the same time. More realistic expectations would aid both implementing and assessing the ACS.

The interaction of multiple processes operating at multiple spatial and temporal scales is difficult to understand, and even more difficult to incorporate into a coherent management strategy. Understanding the relation among different spatial scales is necessary to successfully assess the effects of management policies and activities on aquatic ecosystems in the future. The challenge is to develop a process that not only looks at current aquatic conditions but also:

- Looks broadly to determine the large context.
- Looks historically to assess past trajectories of the systems and natural history.
- Looks ahead to identify potential threats and expectations.

This perspective would allow for a more integrated response to basic questions such as Where are we, where do we want to go, and how do we get there? Watershed assessment is a logical forum to explore these questions.

The failure to recognize the landscape focus of the ACS has precluded consideration of potential options for different management practices and policies. Some practices and policies for managing aquatic ecosystems under the Plan are in many ways similar to those before the Plan. For example, cumulative effects are still determined at the 6th- to 7th-field watershed scale. Thus, management activities are dispersed among watersheds to avoid potential negative effects (fig. 9-4a). But this approach is not necessarily consistent with the landscape focus of the ACS. A potential alternative option was offered by Reeves and others (1995). They suggested that management activities be concentrated in a given watershed for an extended period (fig. 9-4b), rather than dispersed over wider areas. Grant (1990) modeled both scenarios to determine their effects on the pattern of peak flows and found little difference between the two. Concentrating rather than dispersing activities may also confer benefits to terrestrial organisms that require late-successional forests (Franklin and Formann 1987).

Specifying the spatial scale is important when range of natural variation and cumulative effects are discussed or evaluated. At small scales, the HRV is very large, so, except for the most extreme impacts, no cumulative effects may result from management actions. Most assessments of the effects of human activities are made at relatively small scales. Failure to recognize the relation between space and HRV undoubtedly contributed to the current confusion about the ACS and the scales at which it is applied and how compliance is measured.

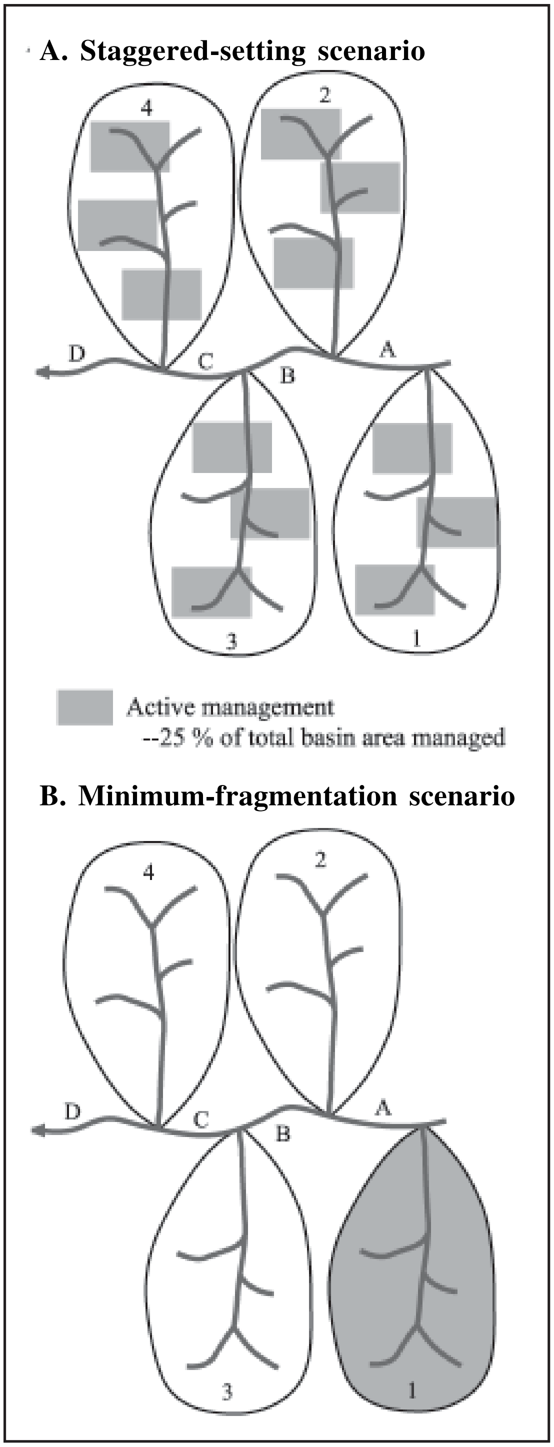


Figure 9-4—Potential approaches to watershed (A) and landscape (B) management. Source: Grant 1990.

The view of aquatic ecosystems as dynamic entities has implications for the network of key watersheds and the potential long-term success of the ACS. First, an underlying assumption about key watersheds was that streams in old-growth forests contained the best habitats for fish. Many of the key watersheds in option 9 of FEMAT (1993) were associated with late-successional reserves. Reeves and others (1995) suggested that streams in mid-successional forests were more productive than those in old-growth forests in the Oregon Coast Range. Whether this pattern is found in other areas is not known at present and could be a future research emphasis. The second implication of treating aquatic ecosystems as dynamic entities deals with the expectations for reserves in dynamic landscapes. Reserves in such a setting cannot be expected to persist for long periods. How future key watersheds will develop and where in the landscape they will occur are key questions for managers, regulators, and researchers to consider.

Riparian Reserves

Ecological functions and distance—

The generalized curves (fig. 9-5) developed in FEMAT (1993) were developed by examining the available scientific literature about key ecological processes in riparian ecosystems. The effects of riparian vegetation decreased with an increasing distance from the streambank (FEMAT 1993). Generally, most ecological processes occurred within 100 feet (about two-thirds the height of a site-potential tree) (fig. 9-5).

An exception was large wood (fig. 9-5a). Large wood provides a crucial ecological function (see Bilby and Bisson 1998, Spence and others 1996) in aquatic ecosystems in the Plan area and is readily acknowledged by land management and regulatory agencies. In developing the generalized curve for wood sources, trees were assumed to reach a stream from a slope distance equal to the height of the tree (FEMAT 1993). Implicit in this assumption, but unstated by FEMAT (1993), was that trees in the riparian zone farthest from the channel would not immediately be

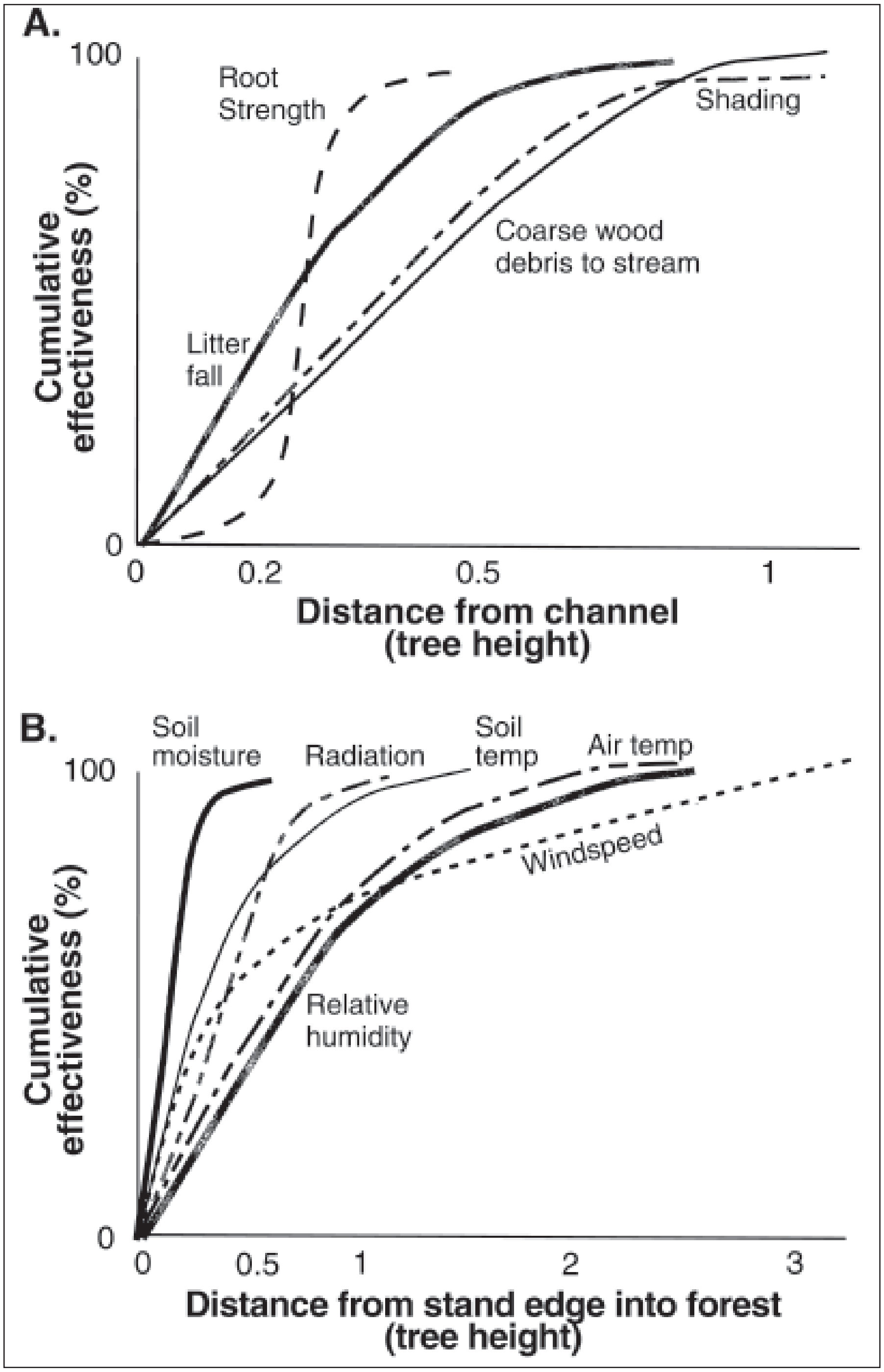


Figure 9-5—Generalized ecological functions in riparian zones as a distance from the stream. Source: FEMAT 1993.

in the current stream channel. These trees could either be recruited over time to the channel or, with wide valley floors, the channel would migrate over time and such pieces could then be in the channel. Bilby and Bisson (1998) noted that the latter process may be an important source of wood for streams in some areas.

Recognition of the role and importance of down wood in riparian areas has increased since the ACS was implemented. Down wood, particularly larger pieces, provides required high-moisture microhabitats for many riparian-associated amphibians (Pilliod and others 2003). It also provides habitat for several species of birds and small mammals found in riparian areas (Kelsey and West 1998). And down wood may collect and impede the movement of finer sediment into streams, preventing fine sediment from reaching streams where it can affect habitat conditions and biota (see references in McIver and Starr 2001, Wondzell and King 2003). This effect may be particularly important in areas where chronic overland erosional processes dominate, which are very rare in the Plan area except after intense fire or severe management disturbance. Trees in the riparian area farthest from the channel are sources of this down wood.

Pete Bisson

Carcasses of salmon and trout provide nutrients for riparian vegetation and a number of aquatic and terrestrial organisms.

Microclimate conditions in riparian areas was another ecological function in addition to wood sources that occurred beyond 100 feet (a distance of about two-thirds of the height of a site potential tree) (fig. 9-5b). Based on the work of Chen (1991), the developers of the ACS (FEMAT 1993) argued wider buffers may be needed to maintain interior microclimatic conditions. Subsequent work by Brosofske and others (1997) supported this contention. Maintaining favorable microhabitat conditions in riparian areas is also important for wildlife species (Kelsey and West 1998).

Headwater streams—

The riparian reserve was one of the cornerstones of the ACS. The riparian reserve network included fish-bearing streams, which had been the focus of management of aquatic ecosystems before FEMAT, as well as small, fishless headwater streams. The latter generally make up 70 percent or more of the stream network (Gomi and others 2002). Before the ACS, these streams were not widely recognized as part of the aquatic ecosystem, but knowledge about and recognition of the ecological importance of headwater streams has increased since then. They are sources of sediment (Benda and Dunne 1997a, 1997b; Zimmerman and Church 2001) and wood (Reeves and others 2003) for fish-bearing streams. They provide habitat for several species of native amphibians (Kelsey and West 1998) and macroinvertebrates (Meyer and Wallace 2001), including recently discovered species (Dieterich and Anderson 2000), and may be important sources of food for fish (Wipfli and Gregovich 2002). Small streams are also storage and processing sites of nutrients and organic matter, important components of the energy base for organisms used by fish for food (Kiffney and others 2000, Wallace and others 1995, Webster and others 1999, Wipfli and Gregovich 2002).

Headwater streams are among the most dynamic portions of the aquatic ecosystems (Naiman and others 1992). Tributary junctions between headwater streams and larger channels are important nodes for regulating material flows in a watershed (Benda and others 2004, Gomi and others 2002) and are the locations where site-scale effects from management activities are often observed. These locations have unique hydrologic, geomorphic, and biological attributes. The movement of sediment, wood, and other materials through these locations results in sites of high biodiversity (Johnson and others 1995, Minshall and others 1985). Habitat in these sites may also range from simple to complex, depending on time from the disturbance (such as landslides and debris flows) and the types and amount of materials delivered to the channel.

Large wood is an important element of stream and river ecosystems. It forms and influences the size and frequency of habitat units for fish and other organisms that depend on aquatic and riparian habitats (Bilby and Bisson 1998, Bilby and Ward 1989, Wallace and others 1995). The size of pieces and amount of wood in the channel also influences the abundance, biomass, and movement of fish (Fausch and Northcote 1992, Harvey and Nakamoto 1998, Harvey and others 1999, Murphy and others 1985, Roni and Quinn 2001). Wood enters streams via chronic and episodic processes (Bisson and others 1987). Chronic processes, such as tree mortality and bank undercutting (Bilby and Bisson 1998, Grette 1985, Murphy and Koski 1989), generally introduce single pieces or relatively small numbers of trees at frequent intervals. Episodic processes usually add large amounts of wood to streams in big but infrequent events, such as windthrow (Harmon and others 1986), wildfire (Agee 1993), severe floods, and landslides and debris flows (Keller and Swanson 1979, May 2002, Reeves and others 2003).

Examinations of wood sources in streams (such as McDade and others 1990, Murphy and Koski 1989, Robison and Beschta 1990) have focused until recently on chronic input from the immediately adjacent riparian zone. Such studies concluded that most of the wood found in streams was derived from within a distance of about 100 feet. Riparian management in forest plans developed before the Plan was based primarily on these cited studies and assumed that most of the wood found in streams came from within 100 feet of the stream. The studies on which this assumption was made, however, either did not consider episodic sources of wood (such as Van Sickle and Gregory 1990) or did not sample study reaches influenced by upslope sources (such as McDade and others 1990). The assumption that all wood came from within 100 feet of the channel based in the cited studies is incorrect, and the potential effectiveness of plans and policies based on it are questionable.

In steep terrain, which is found on much of the Plan area, landslides and debris flows are potentially important mechanisms for delivering sediment and wood from hillslopes and small headwater channels to valley-bottom streams. Reeves and others (2003) found that an estimated 65 percent of the number of pieces and 46 percent of the total volume of wood in a pristine watershed in coastal Oregon came from outside the riparian zone immediately adjacent to the fish-bearing stream. More than 80 percent of the total number of pieces of wood in a western Washington stream (Benda and others 2003b) and a northern California stream (Benda and others 2002) were from upslope sources. Other studies, such as May (2002) and Benda and others (2003a), found large amounts of wood from upslope sources in streams in the Oregon Coast Range and Idaho, respectively.

Pieces of large wood delivered from upslope areas are generally smaller than those originating from the riparian zones along fish-bearing streams. Reeves and others (2003) found that the mean volume of a piece of large wood from upslope areas was one-third the mean size of pieces from stream-adjacent riparian areas in a coastal Oregon stream. Difference in mean size is likely attributable to fire history and other stand-resetting events. Hillslopes are more susceptible to fire and burn more frequently than streamside riparian zones (Agee 1993). Thus, trees in the streamside riparian zone may be disturbed less frequently and achieve larger sizes than upslope trees.

Geomorphic features of a watershed influence the potential contribution of upslope wood sources. Steeper, more highly dissected watersheds will likely have a greater proportion of wood coming from upslope sources than will watersheds with lower gradients. Murphy and Koski (1989) and Martin and Benda (2001) found that upslope sources of wood composed a relatively small proportion of the total wood in streams that they examined in Alaska. The watershed studied by Martin and Benda (2001) had a wide valley floor, so wood was deposited along valley floors away from the main channel. In contrast, Benda and others (2003a) found that wood delivered in landslides after wildfires was deposited in wide valley reaches in the Boise River, Idaho. In a central Oregon coast stream, Reeves and others (2003) found that the amount of upslope-derived wood was greatest in reaches with narrow valley floors.

Even in watersheds where the potential contribution from upslope sources of wood is high, the ability of individual upslope sources to contribute wood to fish-bearing streams can differ widely. Benda and Cundy (1990) identified the features of first-and second-order channels with the greatest potential to deliver sediment and wood to fish-bearing streams in the central Oregon coast. The primary features were gradients of 8 to 10 percent with tributary junction angles <45°. These features can be identified from Digital Elevation Models (DEMs) and topographic maps. Benda and others (N.d.) have developed a process that uses information from DEMs to develop basin-specific information for stratifying landscapes for varying intensity of resource management, identifying ecologically significant terrain for conservation, and prioritizing watershed and instream restoration and monitoring activities.

The presence of large wood from headwater streams influences the behavior of landslides and debris flows and the response of the channel to such events. Large wood in debris flows and landslides influences the runout length of these events (Lancaster and others 2003). Debris flows without wood move faster and longer distances than those with wood, and they are less likely to stop high in the stream network and to reach fish-bearing channels. A debris flow without wood is likely to be primarily a concentrated slurry of sediments of varying sizes that can move at relatively high speeds over long distances scouring substrate and wood from the affected channels. These types of flows are more likely to negatively affect fish-bearing channels rather than have potential favorable effects that result from the presence of wood. They can further delay or impede the development of favorable conditions for fish and other aquatic organisms.

Over time, headwater depressions and channels are filled with material from the surrounding hillslopes, including large wood that falls into these channels, forming obstructions behind which sediments accumulate (Benda and Cundy 1990, May and Gresswell 2004). These areas are evacuated following a landslide or debris flow. This cycle of filling and emptying results in a punctuated movement of sediment and wood to larger, fish-bearing streams (Benda and others 1998), which is—at least, in part—responsible for the long-term productivity of many aquatic ecosystems (Benda and others 2003a, Hogan and others 1998, Reeves and others 1995). The absence of wood to replenish the refilling process may result in a chronic movement of sediment to larger channels, which could lead to those channels developing different characteristics than those that occurred before forest management. Such conditions could be outside the range of watershed conditions to which native biota are adapted (Beschta and others 2004).

Fire and riparian and aquatic ecosystems—
The issue of fire and aquatic ecosystems was given little consideration by the Aquatic Conservation Plan's developers (FEMAT 1993), primarily because the potential threat of fire to aquatic ecosystems was not widely recognized at that time. Since then, numerous studies have examined the effect of fire on upland ecosystems, but relatively few examined aquatic and riparian ecosystems. Those studies that considered riparian areas generally focused on perennial streams, and the specific results differ with geographic location. In general, the frequency and

magnitude (following the definitions of Agee 1993) of fires in riparian areas is less than in adjacent upslope areas. Differences between fire effects on riparian and upland areas are less in regions with more frequent and less severe fires compared to locations where the fire return interval is larger and the fires are more severe. Fire in riparian areas along intermittent streams has not been studied, most likely because the inclusion of these areas as part of the riparian systems is only recently beginning to be recognized. Assuming that the effects of fire on the riparian zones of ephemeral and intermittent streams are similar to fire effects on upland plant communities is probably safe; however, I acknowledge that much additional research is needed.

Wildfire can profoundly affect watersheds and streams and associated aquatic organisms. The immediate effects of severe fires that burn through riparian areas and across small streams may include high mortality or emigration of fishes and other organisms caused by direct heating and changes in water chemistry (Minshall and others 1997, Rieman and Clayton 1997, Spencer and others 2003). Subsequent effects associated with the loss of vegetation and infiltration capacity of soils may include increased erosion, changes in the timing and amount of runoff, elevated stream temperatures and changes in the structure of stream channels (Benda and others 2003a, Wondzell and King 2003). The nature of these changes depends on the extent, continuity, and severity of the fire, and on lithology, landform, and local climate (Luce, in press; Rieman and Clayton 1997; Swanson and others 1988). A severe fire burning through dense fuels can produce extensive areas of hydrophobic soils (DeBano and others 1998). If a large storm follows in steep, highly dissected terrain, the result can be massive erosion and debris or hyper-concentrated flows that completely reorganize entire segments of mountain streams and deposit large volumes of sediment in lower gradient reaches (Benda and others 2003a).

Whether fire is viewed as ecologically catastrophic, however, is a matter of context and scale. Following the Boise fire in central Idaho, most fish populations rebounded quickly, in part through dispersal from unburned stream refugia (Rieman and Clayton 1997). Roughly 10 years after the disturbance, little evidence remains to suggest that the distribution and abundance of fishes in these streams are fundamentally different from similar-sized unburned streams. Beneficial effects of fire, such as increased primary productivity and invertebrate abundances, may offer mechanisms for individual fish to cope with potentially stressful conditions (such as high temperatures) in disturbed streams. Further, on timescales of decades to millennia, large disturbances have been common in these landscapes. Fishes and other species probably evolved mechanisms such as dispersal and plasticity in life history that allow them to recover (Dunham and others 2003, Reeves and others 1995).

Additionally, physical complexity in a stream may increase after a wildfire. Recent work has shown that fire and subsequent hydrologic events can contribute wood and coarse sediment necessary to create and maintain productive instream habitats (Bisson and others 2003, Reeves and others 1995). Benda and others (2003a), for example, have shown how mass erosion and deposition at tributary junctions can produce important heterogeneity in channel structure. Natural disturbances interacting with complex terrain has been linked to a changing mosaic of habitat conditions in both terrestrial and aquatic systems (Bisson and others 2003, Miller and others 2003, Reeves and others 1995). This variation of conditions in space and time may be the key to evolving and maintaining biological diversity and, ultimately, the resilience and productivity of many aquatic populations and communities (Bisson and others 2003, Dunham and others 2003, Poff and Ward 1990).

Land managers may view salvage logging after wildfire as a potential restoration technique by which they can respond to the perceived adverse effects of fire (McIver and Starr 2001). Research on the effects of postfire salvage logging on terrestrial organisms has shown mixed results; some organisms showed no effect, others increased (such as, Blake 1982, Haim and Izhaki 1994), and others declined

(Saab and Dudley 1998). Studies on the potential effects of fire and postfire logging of riparian systems and associated biota are lacking, however. Reeves and others (2006) argue that salvage logging in riparian zones may, among other things, reduce the amount and size of wood delivered to stream channels. This reduction may have immediate and long-term ecological consequences for trophic inputs and physical habitats of streams. Activities associated with salvage logging, including building new roads or opening old ones, may further exacerbate the effects of salvage logging by increasing erosion and fragmentation of the stream network. Although, in some circumstances, concerns about human safety justify salvage logging in a riparian zone, there is presently a paucity of evidence of scientific support for salvage logging in riparian zones (Reeves and others 2006). This certainly is an area worthy of future research.

"Cultural shifts" within the land management agencies—

Implementation of the Plan and ACS bought major changes to the way the affected agencies viewed and managed aquatic resources and watersheds. It is difficult to accurately describe or to quantify these changes, but conversations with agency personnel reveal that the vast majority believe that these changes were the most important effect of the Plan and ACS. The ACS replaced local plans that contained a variety of management directions and objectives with a common framework for managing aquatic and riparian resources on public lands. Additionally, it required a more comprehensive approach to the management of aquatic and riparian resources and much more interaction among disciplines that previously had little interaction. Table 9-3 summarizes these changes in agency culture, analysis, and analytical basis of management. In the view of many of the people responsible for the implementation of the ACS, these changes clearly are the primary successes of the Plan.

In a survey authorized by the Forest Plan Revision Board of Directors of FS Pacific Northwest Region (Region 6), personnel involved with the implementation of the ACS (forest and district fish biologists, hydrologists, and wildlife biologists) believed that ACS was appropriate and that it has led to improved and proactive management of aquatic resources (Heller and others 2004). The respondents also believed that there was a need to develop a single unified regional ACS, and this was accepted by the Board of Directors. A single framework is currently being developed for FS Region 6 with the Plan ACS as its cornerstone.

Summary and Considerations

Producing a quantitative assessment of the ACS of the Plan continues to be challenged by issues of data availability and quality. First, the accuracy and quality of data on some activities is questionable. For example, Baker and others (in press) report in their summary that the FS and BLM reported decommissioning 295 miles of road. When they examined 89 watershed assessments done between 1999 and 2003, they found that road mileage in those watersheds was reduced by 1,179 miles. Data on important indicators of effectiveness, such as miles of streams with water quality problems (that is 303d-listed streams) on federally managed lands and volume of timber harvested in riparian reserves, are not available. Watersheds degraded by management activities before the Plan was implemented were expected to take several years or decades to recover (FEMAT 1993). Thus, it is not too late to assemble credible data on activities and actions done under the auspices of the ACS. Field units are improving watershed conditions by removing and improving roads, in-channel restoration projects, improving riparian areas, and so forth, in addition to providing some timber volume from the riparian reserve network. The land management agencies could consider requiring field units to report uniformly on selected key activities and have the data assembled and accessible in a central location. The availability of such data would allow for at least a more defensible qualitative assessment of the effectiveness of the ACS.

Table 9-3—Changes in paradigms for managing aquatic and riparian resources that occurred as result of the implementation of the Plan and Aquatic Conservation Strategy

Old	New
Management activities can occur unless unacceptable adverse impacts can be shown likely to occur.	Management activities should contribute to, or not retard, attainment of ACS objectives.
There is a variety of individual approaches for the protection and restoration of aquatic and riparian-dependent resources. These are often different between administrative units for no apparent reason.	There is a consistent strategic approach for the protection restoration of aquatic and riparian-dependent resources across the entire Plan area.
Focus is on the condition of individual streams or stream segments or sites. Attention is focused primarily on public land.	Management focus is on process and function of whole watersheds. Special efforts are made to consider and coordinate activities on all ownerships.
Effectiveness monitoring is highly variable between administrative units. Protocols are inconsistent and preclude summarization and analysis across the Plan area.	There is a formal program, with consistent protocols, to monitor effectiveness of the strategy across the Plan area. Data can be summarized and analyzed for the Plan area.
Federal agencies generally work independently. Coordination is often infrequent and driven by "problems." Efforts to involve all stakeholders occur but are not the norm.	The emphasis is to coordinate the activities of federal agencies in the implementation and evaluation of the Plan. Special efforts are made to include all stakeholders.
Proposed actions came from "target" generally unrelated to ecosystem characteristics. Analysis is generally single disciplinary, single scale, and noncollaborative.	There is a multiscale analysis of ecosystem form and function prior to formulating proposed actions.

Source: Heller 2002.

The ACS met its expectation that watershed condition would begin to improve in the first decade of the Plan. The conditions of watersheds in the Plan appear to have improved slightly since the Plan was implemented. The proportion of watersheds whose conditions improved was significantly greater than those that declined. A primary reason for this improvement was an increase in the number of large trees in riparian areas and a decrease in the extent of clearcut harvesting in riparian zones. This general trend of improvement should be expected to continue, and may actually accelerate in the future, if the ACS is implemented in its current form. It is highly likely that these trends would have been the reverse under many of the forest plans that were in place before the ACS.

Science information developed since the Plan was implemented supports the framework and components of the ACS, particularly for the ecological importance of smaller, headwater streams. Also, a growing body of science about the dynamics of aquatic and riparian ecosystems could provide a foundation for developing new management approaches and policies. Scientifically based tools for aiding watershed analysis are also available and could be considered for use by the various agencies.

One of the main topics that could be examined and considered in more detail is that of the relation between spatial scales that are considered by the Plan and the ACS. The Plan and ACS changed the focus of the land management agencies from small spatial scales (i.e., watersheds) to larger scales (that is, landscapes). It appears that the implications of doing this have not been fully recognized or appreciated by the land management or regulatory agencies, and it has created confusion with the public and policymakers. This has precluded the consideration of new options and approaches to management. A rigorous examination of this issue would certainly be worthwhile.

References

Agee, J.K. 1993. Fire ecology of Pacific Northwest forests. Washington, DC: Island Press. 493 p.

Allen, T.F.H.; Hoekstra, T.W. 1992. Toward a unified ecology. New York: Columbia University Press. 384 p.

Baker, D.; Palmer, C.; Tolle, T. [In press]. The Northwest Forest Plan—the first ten years (1994-2003): implementation monitoring: accomplishments and compliance with Plan requirements. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region.

Benda, L.; Miller, D.; Andras, K. [N.d.]. A landscape-riverine terrain characterization and analysis system. 51 p. Unpublished manuscript. On file with: Gordon Reeves, Forest Sciences Laboratory, 3200 SW Jefferson, Corvallis, OR 97331.

Benda, L.; Poff, N.L.; Miller, D.; Dunne, T.; Reeves, G.; Pess, G.; Pollock, M. 2004. The network dynamics hypothesis: how channel networks structure riverine habitats. BioScience. 54: 413-428.

Benda, L.E.; Bigalow, P.; Worsley, W. 2002. Recruitment of in-stream large wood in old-growth and second-growth redwood forests, northern California, U.S.A. Canadian Journal of Forest Research. 32: 1460-1477.

Benda, L.E.; Cundy, T.W. 1990. Predicting deposition of debris flows in mountain channels. Canadian Geotechnical Journal. 27: 409-417.

Benda, L.E.; Dunne, T. 1997a. Stochastic forcing of sediment routing and storage in channel networks. Water Resources Research. 33: 2865-2880.

Benda, L.E.; Dunne, T. 1997b. Stochastic forcing of sediment supply to the channel networks from landsliding and debris flows. Water Resources Research. 33: 2849-2863.

Benda, L.E.; Miller, D.; Bigelow, P.; Andras, K. 2003a. Effects of post-wildfire erosion on channel environments, Boise River, Idaho. Forest Ecology and Management. 178: 105-119.

Benda, L.E.; Miller, D.J.; Dunne, T.; Reeves, G.H.; Agee, J.K. 1998. Dynamic landscape systems. In: Naiman, R.J.; Bilby, R.E., eds. River ecology and management: lessons from the Pacific Coastal ecoregion. New York: Springer: 261-288.

Benda, L.E.; Veldhuisen, C.; Black, J. 2003b. Influence of debris flows on the morphological diversity of channels and valley floor, Olympic Peninsula, Washington. Geological Society of America Bulletin. 115: 1110-1121.

Beschta, R.L.; Rhodes, J.J.; Kauffmann, J.B.; Gresswell, R.E.; Minshall, G.W.; Karr, J.R.; Perry, D.A.; Hauer, F.R.; Frissell, C.A. 2004. Postfire management on forested public lands in the Western United States. Conservation Biology. 18: 957-967.

Bilby, R.E.; Bisson, P.A. 1998. Function and distribution of large woody debris. In: Naiman, R.J.; Bilby, R.E., eds. River ecology and management: lessons from the Pacific Coastal ecoregion. New York: Springer: 324-346.

Bilby, R.E.; Ward, J.W. 1989. Changes in characteristics and functions of large woody debris with increasing size of streams in southwestern Washington. Transactions of the American Fisheries Society. 118: 368-378.

Bisson, P.A.; Bilby, R.E.; Byrant, M.D.; Dolloff, C.A.; Grette, G.B.; House, R.A.; Murphy, M.L.; Koski, K.V.; Sedell, J.R. 1987. Large woody debris in forested streams in the Pacific Northwest: past, present, and future. In: Salo, E.O.; Cundy, T.W., eds. Streamside management and fishery interactions. Seattle, WA: Institute of Forest Resources, University of Washington: 143-190.

Bisson, P.A.; Nielson, J.L.; Palmason, R.A.; Grove, L.E. 1982. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low stream flow. In: Armantrout, N.B., ed. Acquisition and utilization of aquatic habitat inventory information, symposium proceedings. Portland, OR: Western Division, American Fisheries Society: 62-73.

Bisson, P.A.; Rieman, B.E.; Luce, C.; Hessburg, P.F.; Lee, D.C.; Kershner, J.L.; Reeves, G.H.; Gresswell, R.E. 2003. Fire and aquatic ecosystems of the Western USA: current knowledge and key questions. Forest Ecology and Management. 178: 213-229.

Blake, J.G. 1982. Influence of fire and logging on nonbreeding bird communities of ponderosa pine forests. Journal of Wildlife Management. 46: 404-415.

Brosofske, K.D.; Chen, J.; Naiman, R.J.; Franklin, J.F. 1997. Harvesting effects on microclimatic gradients from small streams to uplands in western Washington. Ecological Applications. 7: 1188-1200.

Buffington, J.M.; Lisle, T.E.; Woodsmith, R.D.; Hilton, S. 2002. Controls on the size and occurrence of pools in coarse-grained forest rivers. River Research and Applications. 18: 507-531.

Burnett, K. 2004. Personal communication. Research fisheries biologist, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson, Corvallis, OR 97331.

Burnett, K.; Reeves, G.; Miller, D.; Clarke, S.; Christiansen, K.; Vance-Borland, K. 2003. A first step toward broad-scale identification of freshwater protected areas for Pacific salmon and trout in Oregon, U.S.A. In: Beumer, J.P.; Grant, A.; Smith, D.C., eds. Protected areas: What works best and how do we know? Proceedings of the World Congress on aquatic protected areas. Cairns, Australia: Australian Society for Fish Biology: 144-154.

Burnett, K.M. 2001. Relationships among juvenile anadromous salmonids, their freshwater habitat, and landscape characteristics over multiple years and spatial scales in Elk River, Oregon. Corvallis, OR: Oregon State University. 244 p. Ph.D. dissertation.

Burnett, K.M.; Reeves, G.H.; Miller, D.J.; Clarke, S.E.; Vance-Borland, K.; Christiansen, K.R. [In press]. Distribution of salmon habitat potential relative to landscape characteristics and implications for conservation. Ecological Applications.

Chen, J. 1991. Edge effects: microclimatic pattern and biological responses in old-growth Douglas-fir forests. Seattle, WA: University of Washington. 174 p. Ph.D. dissertation.

Cissel, J.H.; Swanson, F.J.; Grant, G.E.; Olson, D.H.; Gregory, S.V.; Garmen, S.L.; Ashkenas, L.R.; Hunter, M.G.; Kertis, J.A.; Mayo, J.H.; McSwain, M.D.; Sweetland, S.G.; Swindle, K.A.; Wallin, D.O. 1998. A landscape plan based on historical fire regimes for a managed forest ecosystem: the Augusta Creek study. Gen. Tech. Rep. PNW-GTR 422. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 82 p.

Concannon, J.A.; Shafer, C.L.; DeVelice, R.L.; Sauvajot, R.M.; Boudreau, S.L.; Demeo, T.E.; Dryden, J. 1999. Describing landscape diversity: a fundamental tool for landscape management. In: Szaro, R.C.; Johnson, N.C.; Sexton, W.T.; Malick, A.J., eds. Ecological stewardship: a common reference for ecosystem management. Oxford, United Kingdom: Elsevier Science, Ltd.: 195-218. Vol. 2.

DeBano, L.F.; Neary, D.G.; Elliott. P.F. 1998. Fire's effect on ecosystems. New York: Willey. 333 p.

Dieterich, M.; Anderson, N.H. 2000. The invertebrate fauna of summer-dry streams in western Oregon. Archive für Hydrobiologie. 147: 273-295.

Dolloff, C.A.; Flebbe, P.A.; Owen, M.W. 1994. Fish habitat and fish populations in a southern Appalachian watershed before and after Hurrican Hugo. Transactions of the American Fisheries Society. 123: 668-678.

Dunham, J.B.; Young, M.; Gresswell, R.; Rieman, B.E. 2003. Effects of fire on fish populations: landscape perspectives on persistence of native fishes and non-native fish invasions. Forest Ecology and Management. 178: 183-196.

Everest, F.H.; Reeves, G.H. [In press]. Riparian and aquatic habitats in the Pacific Northwest and southeast Alaska: ecology, management history, and potential management strategies. Gen. Tech. Rep. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Fausch, K.D.; Northcote, T.G. 1992. Large woody debris and salmonid habitat in a small coastal British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences. 49: 682-693.

Fausch, K.D.; Torgersen, C.E.; Baxter, C.V.; Li, H.W. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. BioScience. 52: 483-498.

Fisher, S.G. 1997. Creativity, idea generation, and the functional morphology of streams. Journal of the North American Benthological Society. 16: 305-318.

Forest Ecosystem Management Assessment Team [FEMAT]. 1993. Forest ecosystem management: an ecological, economic, and social assessment. Portland, OR: U.S. Department of Agriculture; U.S. Department of the Interior [and others]. [Irregular pagination].

Franklin, J.F.; Formann, R.T.T. 1987. Creating landscape pattern by forest cutting: ecological consequences and principles. Landscape Ecology. 1: 5-18.

Frissell, C.A.; Liss, W.J.; Warren, C.E.; Hurley, M.D. 1986. A hierarchial framework for stream habitat classification: viewing streams in a watershed context. Environmental Management. 10: 199-214.

Fulton, J.D.; LaBrasseur, R.J. 1985. Interannual shifting of the Subarctic Boundary and some of the biotic effects on juvenile salmonids. In: Wooster, W.S.; Fluharty, D.L., eds. El Niño effects in the eastern subarctic Pacific. Seattle, WA: Washington Sea Grant Program: 237-247.

Furniss, M.J.; Roelofs, T.D.; Yee, C.S. 1991. Road construction and maintenance. American Fisheries Society Special Publication. 19: 297-324.

Gallo, K.; Lanigan, S.H.; Eldred, P.; Gordon, S.N.; Moyer, C. 2005. Northwest Forest Plan—the first 10 years (1994-2003): preliminary assessment of the condition of watersheds. Gen. Tech. Rep. PNW-GTR-647. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 133 p.

Gomi, T.; Sidle, R.C.; Richardson, J.S. 2002. Understanding processes and downstream linkages of headwater streams. BioScience. 52: 905-916.

Gosz, J.R.; Asher, J.; Holder, B.; Knight, R.; Naiman, R.; Raines, G.; Stine, P.; Wigley, T.B. 1999. An ecosystem approach for understanding landscape diversity. In: Szaro, R.C.; Johnson, N.C.; Sexton, W.T.; Malick, A.J., eds. Ecological stewardship: a common reference for ecosystem management. Oxford, United Kingdom: Elsevier Science, Ltd.: 157-194. Vol. 2.

Grant, G.E. 1990. Hydrologic, geomorphic, and aquatic habitat implications for old and new forestry. In: Pearson, A.F.; Challenger, D.A., eds. Proceedings of the symposium forest managed and wild: differences and consequences. Vancouver, British Columbia: University of British Columbia: 35-53.

Gresswell, R.E. 1999. Fire and aquatic ecosystems in forested biomes of North America. Transactions of the American Fisheries Society. 128: 193-221.

Grette, G.B. 1985. The abundance and role of large organic debris in juvenile salmonid habitat in streams in second growth and unlogged forests. Seattle, WA: University of Washington. 105 p. M.S. thesis.

Haim, A.; Izhaki, I. 1994. Changes in rodent community during recovery from fire: relevance to conservation. Biodiversity Conservation. 3: 573-585.

Hansen, A.J.; Urban, D.L. 1992. Avian response to landscape patterns: the role of species life histories. Landscape Ecology. 7: 163-180.

Harmon, M.E.; Franklin, J.F.; Swanson, F.J.; Sollins, P.; Gregory, S.V.; Lattin, J.D.; Anderson, N.H.; Cline, S.P.; Aumen, N.G.; Sedell, J.R.; Lienkaemper, G.W.; Cromack, K., Jr.; Cummins, K.W. 1986. Ecology of coarse woody debris in temperate streams. Advances in Ecology. 15: 133-302.

Harrison, S.; Quinn, J.F. 1989. Correlated environments and the persistence of metapopulations. Oikos. 56: 293-298.

Harvey, B.C.; Nakamoto, R.J. 1998. The influence of large wood debris on retention, immigration, and growth of coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) in stream pools. Canadian Journal of Fisheries and Aquatic Sciences. 55: 1902-1908.

Harvey, B.C.; Nakamoto, R.J.; White, I.L. 1999. Influence of large woody debris and bankfull flood on movement of adult resident coastal trout (*Oncorhynchus clarki*) during fall and winter. Canandian Journal of Fisheries and Aquatic Sciences. 56: 2161-2166.

Heller, D. 2002. A new paradigm for salmon and watershed restoration. 7 p. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, Natural Resources, P.O. Box 3890, Portland, OR 97208.

Heller, D.; McCammon, B.; Roper, B. 2004. Aquatic strategies in Region 6–Are they working? 6 p. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, Natural Resources, P.O. Box 3890, Portland, OR 97208.

Hobbs, R.J.; Huenneke, L.F. 1992. Disturbance, diversity, and invasion: implications for conservation. Conservation Biology. 6: 324-337.

Hogan, D.L.; Bird, S.A.; Rice, S. 1998. Stream channel morphology and recovery processes. In: Hogan, D.L.; Tschaplinski, P.J.; Chatwin, S., eds. Carnation Creek and Queen Charlotte Islands fish/forestry workshop: applying 20 years of coast research to management solutions. Land management handbook. Victoria, BC: Crown Publications, Inc.: 77-96.

Holling, C.S. 1992. Cross-scale morphology, geometry, and dynamics of ecosystems. Ecological Monographs. 62: 447-502.

Independent Multidisciplinary Scientific Team. 1999. Recovery of wild salmonids in western Oregon forests: Oregon Forest Practices Act rules and the measures in the Oregon Plan for salmon and watersheds. Tech. Rep. 1999-1 to the Oregon Plan for Salmon and Watersheds. Salem, OR: Governor's Natural Resource Office. 99 p.

Johnson, B.L.; Richardson, W.B.; Naimo, T.J. 1995. Past, present, and future concepts in large river ecology. BioScience. 45: 134-141.

Johnson, K.N.; Franklin, J.F.; Thomas, J.W.; Gordon, J. 1991. Alternatives for management of late-successional forests of the Pacific Northwest. A report to the Agriculture Committee and the Merchant Marine Fisheries Committee of the U.S. House of Representatives. 59 p. http://pnwin.nbii.gov/nwfp/alternatives.pdf. (February 2006).

Keane, R.E.; Parsons, R.A.; Hessberg, P.F. 2002. Estimating historical range and variation of landscape patch dynamics: limitations of the simulation approach. Ecological Modeling. 151: 29-49.

Keller, E.A.; Swanson, F.J. 1979. Effects of large organic material on channel form and fluvial processes. Earth Surface Processes. 4: 361-380.

Kelsey, K.A.; West, S.D. 1998. Riparian wildlife. In: Naiman, R.J.; Bilby, R.E., eds. River ecology and management: lessons from the Pacific coastal ecoregion. New York: Springer-Verlag: 235-260.

Kiffney, P.M.; Richardson, J.S.; Feller, M.C. 2000. Fluvial and epilithic organic material dynamics of headwater streams of southwestern British Columbia, Canada. Archive für Hydrobiologie. 148: 109-129.

Lancaster, S.T.; Hayes, S.K.; Grant, G.E. 2003. Effects of wood on debris flow runout in small mountain watersheds. Water Resources Research. 39: 1168.

Landres, P.B.; Morgan, P.; Swanson, F.J. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications. 9: 1179-1188.

Lawson, P.W. 1993. Cycles in ocean productivity, trends in habitat quality, and restoration of salmon runs in Oregon. Fisheries. 18(8): 6-10.

Levin, S. 1974. Dispersion and population interactions. American Naturalist. 108: 207-228.

Lindenmayer, D.B.; Franklin, J.F. 2002. Conserving forest biodiversity: a comprehensive multiscale approach. Washington, DC: Island Press. 351 p.

Luce, C.H. [In press]. Fire effects on runoff generation processes. In: Anderson, M.G., ed. Encyclopedia of hydrological sciences. Chichester, England: John Wiley and Sons.

Lugo, A.E.; Baron, J.S.; Frost, T.P.; Cundy, T.W.; Dittberner, P. 1999. Ecosystem processes and functioning. In: Szaro, R.C.; Johnson, N.C.; Sexton, W.T.; Malick, A.J., eds. Ecological stewardship: a common reference for ecosystem management. Oxford, United Kingdom: Elsevier Science, Ltd.: 219-254. Vol. 2.

Mantua, N.J.; Hare, S.R.; Zhang, Y.; Wallace, J.M.; Francis, R.C. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the American Meterological Society. 78: 1069-1079.

Martin, D.J.; Benda, L.E. 2001. Patterns of instream wood recruitment and transport at the watershed scale. Transactions of the American Fisheries Society. 130: 940-958.

May, C.L. 2002. Debris flows through different forest age classes in the central Oregon Coast Range. Journal of the American Water Resources Association. 38: 1-17.

May, C.L.; Gresswell, R.E. 2004. Processes and rates of sediment and wood accumulation in headwater streams of the central Oregon Coast Range. Earth Surface Processes and Landforms. 28: 409-424.

McDade, M.H.; Swanson, F.J.; McKee, W.A.; Franklin, J.F.; Van Sickle, J. 1990. Source distance of coarse woody debris entering small streams in western Oregon and Washington. Canadian Journal of Forest Research. 20: 326-330.

McIver, J.D.; Starr, L. 2001. A literature review on the environmental effects of postfire logging. Western Journal of Applied Forestry. 16: 159-168.

Megahan, W.F.; Kidd, W.J. 1972. Effects of logging and logging roads on erosion and sediment deposition from steep terrain. Journal of Forestry. 70: 136-171.

Meyer, J.L.; Wallace, J.B. 2001. Lost linkages and lotic ecology: rediscovering small streams. In: Press, M.C.; Huntley, N.J.; Levins, S., eds. Ecology: achievement and challenge. Oxford, United Kingdom: Blackwell Scientific: 295-317.

Miller, D.; Luce, C.H.; Benda, L.E. 2003. Time, space, and episodicity of physical disturbance in streams. Forest Ecology and Management. 178: 121-140.

Minshall, G.W.; Cummins, K.W.; Peterson, R.C.; Cushing, C.E.; Bruns, D.A.; Sedell, J.R.; Vannote, R.L. 1985. Development in stream ecosystem theory. Canadian Journal of Fisheries and Aquatic Sciences. 42: 1045-1055.

Minshall, G.W.; Robison, C.T.; Lawrence, D.E. 1997. Postfire responses of lotic ecosystems in Yellowstone National Park, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences. 54: 2509-2525.

Moeur, M.; Spies, T.A.; Hemstrom, M.; Alegria, J.; Browning, J.; Cissel, J.; Cohen, W.B.; Demeo, T.E.; Healey, S.; Warbington, R. 2005. Northwest Forest Plan–the first 10 years (1994-2003): status and trends of late-successional and old-growth forest. Gen. Tech. Rep. PNW-GTR-646. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 142 p.

Montgomery, D.R.; Buffington, J.M. 1993. Channel classification, prediction of channel response, and assessment of channel condition. Washington State Timber, Fish and Wildlife Tech. Rep. TFW-SH10-93-002. Seattle, WA: University of Washington. 84 p. [plus maps].

Montgomery, D.R.; Grant, G.E.; Sullivan, K. 1995. Watershed analysis as a framework for implementing ecosystem management. Water Resources Bulletin. 31: 369-386.

Montgomery, D.R.; Massong, T.M.; Hawley, S.C.S. 2003. Influence of debris flows and log jams on the location of pools and alluvial channel reaches, Oregon Coast Range. Geological Society of America Bulletin. 115: 78-88.

Moyle, P.B.; Sato, G.M. 1991. On the design of preserves to protect native fish. In: Minckley, W.L.; Deacon, J.E., eds. Battle against extinction: native fish management in the American West. Tucson, AZ: University of Arizona Press: 155-169.

Murphy, M.L.; Koski, K.V. 1989. Input and depletion of coarse woody debris in Alaska streams. North American Journal of Fisheries Management. 9: 427-436.

Murphy, M.L.; Koski, K.V.; Heifetz, J.; Johnson, S.W.; Kirchofer, D.; Thedinga, J.F. 1985. Role of large organic debris as winter habitat for juvenile salmonids in Alaska streams. Proceedings Western Association of Fish and Wildlife Agencies. 1984: 251-262.

Naiman, R.J.; Beechie, T.J.; Benda, L.E.; Berg, D.R.; Bisson, P.A.; McDonald, L.H.; O'Conner, M.D.; Olson, P.L.; Steel, E.A. 1992. Fundamental elements of ecologically healthy watersheds in the Pacific Northwest coastal ecoregion. In: Naiman, R.J., ed. Watershed management: balancing sustainability and environmental change. New York: Springer-Verlag: 127-188.

Naiman, R.J.; Bilby, R.E., eds. 1998. River ecology and management: lessons from the Pacific coastal ecoregion. New York: Springer-Verlag. 705 p.

National Research Council. 1996. Upstream: salmon and society in the Pacific Northwest. Washington, DC: National Academy Press. 452 p.

Newbold, J.D.; Elwood, J.W.; O'Neill, R.V.; Van Winkle, W. 1982. Nutrient spiraling in streams: implications for nutrient limitation and invertebrate activities. American Naturalist. 120: 628-652.

Nickelson, T.E.; Rodgers, J.D.; Johnson, S.L.; Solazzi, M.F. 1992. Seasonal changes in habitat use by juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams. Canadian Journal of Fisheries and Aquatic Sciences. 43: 527-535.

O'Neill, R.V.; DeAngelis, D.L.; Waide, J.B.; Allen, T.F.H. 1986. A hierarchical concept of ecosystems. Monographs in Population Biology 23. Princeton, NJ: Princeton University Press. 253 p.

Overton, W.S. 1977. A strategy for model construction. In: Hall, C.A.S.; Day, J.W., eds. Ecosystem modeling in theory and practice: an introduction with case histories. New York: John Wiley and Sons: 50-73.

Peets, S.; Doelker, A. 2005. Northwest Forest Plan Aquatic Conservation Strategy: the implementation of the Northwest Forest Plan Aquatic Conservation Strategy on BLM and FS-administered lands within the Oregon Coastal Coho ESU. Unpublished report. 43 p. On file with: S. Peets, Siuslaw National Forest, 4077 SW Research Way, P.O. Box 1148, Corvallis, OR 97339.

Pilliod, D.S.; Bury, R.B.; Hyde, E.J.; Pearl, C.A.; Corn, P.S. 2003. Fire and amphibians in North America. Forest Ecology and Management. 178: 163-181.

Poff, N.L.; Ward, J.V. 1990. Physical habitat template of lotic ecosystems: recovery in the context of spatial heterogeneity. Environmental Management. 14: 629-645.

Potondy, J.P.; Cole, G.F.; Megahan, W.F. 1991. A procedure for estimating sediment yields from forested watersheds. In: Proceedings: fifth federal interagency sedimentation conference. Washington, DC: Federal Energy Regulatory Commission: 12-46—12-54.

Quigley, T.M.; Arbelbide, S.J., tech. eds. 1997. An assessment of ecosystem components in the interior Columbia and portions of the Klamath and the Great Basin. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. Vol. 1-4.

Reeves, G.H.; Benda, L.E.; Burnett, K.M.; Bisson, P.A.; Sedell, J.R. 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. In: Nielsen, J., ed. Evolution in the aquatic ecosystem: defining unique units in population conservation. Bethesda, MD: American Fisheries Society Symposium: 17: 334-349.

Reeves, G.H.; Bisson, P.A.; Dambacher, J.M. 1998. Fish communities. In: Naiman, R.J.; Bilby, R.E., eds. River ecology and management: lessons from the Pacific coastal ecoregion. New York: Springer-Verlag: 200-234.

Reeves, G.H.; Burnett, K.M.; Gregory, S.V. 2002. Fish and aquatic ecosystems in the Oregon Coast Range. In: Hobbs, S.D.; Hayes, J.P.; Johnson, R.L.; Reeves, G.H.; Spies, T.A.; Tappeiner, J.C., II; Wells, G.E., eds. Forest and stream management in the Oregon Coast Range. Corvallis, OR: Oregon State University Press: 68-98.

Reeves, G.H.; Burnett, K.M.; McGarry, E.V. 2003. Sources of large wood in a pristine watershed in coastal Oregon. Canadian Journal of Forest Research. 33: 1363-1370.

Reeves, G.H.; Everest, F.H.; Sedell, J.R. 1993. Diversity of juvenile anadromous salmonid assemblages in coastal Oregon basins with different levels of timber harvest. Transactions of the American Fisheries Society. 122: 309-317.

Reeves, G.H.; Hohler, D.B.; Larsen, D.P.; Busch, D.E.; Kratz, K.; Reynolds, K.; Stein, K.F.; Atzet, T.; Hays, P.; Tehan, M. 2004. Effectiveness monitoring for the aquatic and riparian component of the Northwest Forest Plan: conceptual framework and options. Gen. Tech. Rep. PNW-GTR-577. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 71 p.

Reeves, G.H.; Rieman, B.E.; Bisson, P.A.; Benda, L.E. 2006. Salvage logging in riparian ecosystems. Conservation Biology. 20: 994-1004.

Reid, L.M. 1998. Watershed analysis. In: Naiman, R.J.; Bilby, R.E., eds. River ecology and management: lessons from the Pacific coastal ecoregion. New York: Springer-Verlag: 476-501.

Resh, V.H.; Brown, A.V.; Covich, A.P.; Gurtz, M.E.; Li, H.W.; Minshall, G.W.; Reice, S.R.; Sheldon, A.L.; Wallace, J.B.; Wissmar, R. 1988. The role of disturbance in stream ecology. Journal of the North American Benthological Society. 7: 433-455.

Rieman, B.E.; Clayton, J. 1997. Fire and fish: issues of forest health and conservation of native fishes. Fisheries. 22: 6-15.

Robison, E.G.; Beschta, R.L. 1990. Characteristics of coarse woody debris for several coastal streams of southeast Alaska, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences. 47: 1684-1693.

Roghair, C.N.; Dolloff, C.A.; Underwood, M.T. 2002. Response of a brook trout population and instream habitat to catastrophic flood and debris flow. Transactions of the American Fisheries Society. 131: 718-730.

Roni, P.; Quinn, T.P. 2001. Density and size of juvenile salmonid in response to placement of large woody debris in western Oregon and Washington. Canadian Journal of Fisheries and Aquatic Sciences. 58: 282-292.

Rosgen, D.L. 1994. A classification of natural rivers. Catena. 22: 169-199.

Saab, V.; Dudley, J. 1998. Responses of cavity-nesting birds to stand-replacement fire and salvage logging in pondersoa pine/Douglas-fir forests of southwestern Idaho. Res. Pap. RMRS-RP-11. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 17 p.

Sedell, J.R.; Reeves, G.H.; Hauer, F.R.; Stanford, J.A.; Hawkins, C.P. 1990. Role of refugia in recovery from disturbances: modern fragmented and disconnected landscapes. Environmental Management. 14: 711-724.

Sokal, R.R.; Rohlf, F.J. 1969. Biometry. San Francisco: W.H. Freeman and Co. 749 p.

Spence, B.C.; Lomnicky, G.A.; Hughes, R.M.; Novitzki, R.P. 1996. An ecosystem approach to salmonid conservation. TR-4501-96-6057. Corvallis, OR: ManTech Environmental Reserch Services Corp. 356 p.

Spencer, C.N.; Gabel, K.O.; Howard, F.R. 2003. Wildfire effects on stream food webs and nutrient dynamics in Glacier National Park, USA. Forest Ecology and Management. 178: 5-22.

Spies, T.A; McComb, B.C.; Kennedy, R.; McGrath, M.T.; Olsen, K.; Pabst, R.J. [In press]. Potential effects of forest policies on terrestrial biodiversity in a multi-ownership province. Ecological Applications.

Swanson, F.J.; Kratz, T.K.; Caine, N.; Woodmansee, R.G. 1988. Landform effects on ecosystem patterns and processes. BioScience. 38: 92-98.

Thomas, J.W.; Raphael, M.G.; Anthony, R.G.; Forsman, E.D.; Gunderson, A.G.; Holthausen, R.S.; Marcot, B.G.; Reeves, G.H.; Sedell, J.R.; Solis, D.M. 1993. Viability assessments and management considerations for species associated with late-successional and old-growth forests of the Pacific Northwest. Washington, DC: U.S. Department of Agriculture, Forest Service, U.S. Government Printing Office. 530 p.

Trombulak, S.C.; Frissell, C.A. 1999. Review of the ecological effects of roads on terrestrial and aquatic communities. Conservation Biology. 14: 18-30.

Turner, M.G.; Romme, W.H.; Gardner, R.H. 1994. Landscape disturbance models and the long-term dynamics of natural areas. Natural Areas Journal. 14: 3-11.

U.S. Department of Agriculture. 1992. Background report for development of the Forest Service management strategy for Pacific salmon and steelhead habitat. Washington, DC: 41 p.

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management [USDA and USDI]. 1994. Record of decision for amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl. Standards and guidelines for management of habitat for late-successional and old-growth forest related species within the range of the northern spotted owl. [Place of publication unknown]. 74 p.

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management [USDA and USDI]. 2003. Final supplemental environmental impact statement: clarification of language in the 1994 record of decision for the Northwest Forest Plan national forests and Bureau of Land Management districts within the range of the northern spotted owl. Portland, OR. 100 p.

Van Sickle, J.; Gregory, S.V. 1990. Modeling inputs of large woody debris to streams from falling trees. Canadian Journal of Forest Research. 20: 1593-1601.

Vannote, R.L.; Minshall, G.W.; Cummins, K.W.; Sedell, J.R.; Cushing, C.E. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences. 37: 130-137.

Wallace, J.B.; Webster, J.R.; Meyer, J.L. 1995. Influence of log additions on physical and biotic characteristics of a mountain stream. Canadian Journal of Fisheries and Aquatic Sciences. 52: 2120-2137.

Washington Forest Practices Board. 1993. Standard methodology for conducting watershed analysis. Version 1.2. 49 p.

Webster, J.R.; Benfield, E.F.; Ehrman, T.P.; Schaeffer, M.A.; Tank, J.L.; Hutchens, J.J.; D'Angelo, D.J. 1999. What happens to allochthonous material that falls into streams? A synthesis of new and published information from Coweeta. Freshwater Biology. 41: 687-705.

Wemple, B.C.; Swanson, F.J.; Jones, J.A. 2001. Forest roads and geomorphic process interactions, Cascade Range, Oregon. Earth Surface Processes and Landforms. 26: 191-204.

White, P.S.; Pickett, S.T.A. 1985. Natural disturbance and patch dynamics: an introduction. In: Pickett, S.T.A.; White, P.S., eds. The ecology of natural disturbance and patch dynamics. Orlando, FL: Academic Press: 3-13.

Williams, J.E.; Johnson, J.E.; Henderson, D.A.; Contreras-Balderas, S.; Williams, J.D.; Navarro-Mendoza, M.; McCallister, D.E.; Deacon, J.E. 1989. Fishes of North America, endangered, threatened, and of special concern. Fisheries. 14(6): 2-21.

Wimberly, M.C.; Spies, T.A.; Long, C.J.; Whitlock, C. 2000. Simulating historical variability in the Oregon Coast Range. Conservation Biology. 14: 167-180.

Wipfli, M.S.; Gregovich, D.P. 2002. Invertebrates and detritus export from fishless headwater streams in southeastern Alaska: Implications for downstream salmonid populations. Freshwater Biology. 47: 957-970.

Wondzell, S.M.; King, J. 2003. Postfire erosional processes in the Pacific Northwest and Rocky Mountain regions. Forest Ecology and Management. 178: 75-87.

Young, M.K.; Hubert, W.A.; Wesche, T.A. 1991. Selection of measures of substrate composition to estimate survival to emergence of salmonids and to detect changes in stream substrates. North American Journal of Fisheries Management. 11: 339-346.

Zimmerman, A.; Church, M. 2001. Channel morphology, gradient profiles and bed stresses during flood in a step-pool channel. Geomorphology. 40: 311-327.