Wildfire and Native Fish: Issues of Forest He of Sensitive Species

By Bruce Rieman and Jim Clayton

ABSTRACT

Issues related to forest health and the threat of larger, more destructive wildfires have led to major new initiatives to restructure and recompose forest communities in the western United States. Proposed solutions will depend, in part, on silvicultural treatments and prescribed burning. Large fires can produce dramatic changes in aquatic systems, including altered sediment and flow regimes, fish mortality, and even local extinctions. Responses of salmonid populations to large disturbances such as fire indicate that complexity and spatial diversity of habitats are important to the resilience and persistence of populations. Some populations retain the ecological diversity necessary to persist in the face of large fires, and natural events such as wildfire have been important in creating and maintaining habitat diversity. Although timber harvest and fire can precipitate similar changes in watershed processes, we do not necessarily expect the physical and ecological consequences of large fires and timber harvest to be the same. We agree that healthy forests are fundamental to healthy aquatic ecosystems. In their haste to restore unhealthy forests, however, managers must take care to avoid simplistic solutions that compound problems already present in the management of aquatic ecosystems and native fishes. Management to restore ecological structure, composition, and process is largely experimental and potentially risky. We propose that the mosaic of conditions in both terrestrial and aquatic systems provides an opportunity to learn and adapt new management without placing key remnant aquatic habitats and populations at risk.

Some concerned scientists and managers have suggested that the increasing risk of uncharacteristically large or damaging wildfires in forests across the western United States may threaten the integrity of whole landscapes. In response, major initiatives to correct such problems have been proposed, some of which rely largely on silvicultural activities such as increased timber harvest, thinning, and prescribed fire. Do these and associated activities represent higher or lower risks for native fishes and aquatic environments?

Wildfires and forest health have dominated much of the discussion and interest regarding management of forested lands in the West in recent years. It is increasingly evident that forested landscapes throughout the region have been dramatically altered by past land management activities. Selective and extensive timber harvest, silviculture, fire suppression, and grazing practices have substantially changed the structure and composition of forest communities (Franklin 1992; Lehmkuhl et al. 1994; Hessburg et al. 1997). The general decline in abundance of large, old trees and old forest (Henjum et al. 1994; Hessburg et al. 1997) is a familiar issue. But ecologists are now recognizing that changes in structure and composition are symptomatic of more fundamental changes in patterns of disturbance and other ecological processes. Forests that were once mosaics of species, ages, and patterns in crown cover have been simplified. Many are now dominated by higher-density, middle-aged stands that are more vulnerable to pest infestations and fire (Huff et al. 1995; Hessburg et al. 1997; Hann et al. 1997). The more homogeneous patterns in vegetation and the increased fuel loadings are thought to increase landscape vulnerability to larger stand-replacement fires (Agee 1988, 1994; Huff et al. 1995). Large fires burning throughout the West in recent years may have been the result of these changes (e.g., Christensen et al. 1989; Barbouletos and Morelan 1995), although climatic patterns may be important as well (e.g., Johnson et al. 1995, 1996).

The possibility of uncharacteristically large, damaging fires and declining forest health holds important implications for land managers. Larger fires mean lost timber values, increased threats to private property on an expanding urban-wild-land interface, and higher costs associated with fire suppression. Uncharacteristic fires may also threaten properties of ecosystems. Direct effects of intense fires and subsequent impacts on hydrologic regimes, erosion, debris flows, woody debris, and riparian cover can strongly influence the structure and function of aquatic systems (Swanson 1981; Megahan 1991; Bozek and Young 1994). Intense fires and the associated environmental responses (e.g., elevated stream temperatures, large sediment pulses, and debris flows) may

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result in direct mortality of fish (Minshall and Brock 1991; Bozek and Young 1994; Rinne 1996) and other aquatic organisms (Rinne 1996), and even extinction of local populations (Propst et al. 1992; Rinne 1996).

Concern about such severe possibilities has focused interest on active vegetation management, galvanizing efforts to actively restructure or recompose forest communities (Barbouletos and Morelan 1995; see other papers in Eskew 1995). Managers have justified these initiatives by citing the spectrum of potential social, economic, and ecological costs. For example, some managers have argued that changing fire patterns represent one of the most important threats to the persistence of native, threatened, endangered, or sensitive species such as bull trout (*Salvelinus confluentus*), chinook salmon (*Oncorhynchus tshawytscha*), and redband/rainbow trout (*O. mykiss*). Given the general decline of many native fishes throughout the West (Williams et al. 1989; Nehlsen et al. 1991; Lee et al. 1997) and the seemingly pervasive nature of forest health problems, it may be prudent to proceed quickly with active vegetation management in all "unhealthy" forests. But such management carries costs and risks that also must be considered.

Forest health and ecosystem management initiatives seek to recreate a structure and composition in forest communities that is more consistent with natural disturbance regimes. Although prescribed and natural fires can play an important role (Carlson et al. 1995; Huff et al. 1995), silvicultural activities that include thinning and harvest will be desirable where feasible. Although wildfire and timber harvest both can result in changes to watershed processes that conceptually are similar (e.g., increased surface erosion and water yield), they also may differ fundamentally. Disturbance by fire affects ecosystems in complex ways, but activities that reduce the risk of fire, insect, or pathogen outbreaks by changing vegetation structure and composition, and fuel management also lead to complicated effects. Mimicking natural disturbance may not be as easy as hoped.

We believe that managers often hold overly simplistic views of the forest health problem. In our experience, some believe that forest health can be improved simply by managing vegetation through silvicultural treatments that manipulate structure and composition. While this viewpoint is attractive because it provides some clear direction for managers, key elements of large ecosystems such as soil and watershed processes can be discounted. Land management activities influence many ecosystem components, and clearly more than the forests have changed. In other cases managers recognize that forest attributes are part of a larger ecosystem, but still seem to imply that treating symptoms (terrestrial vegetation) will lead to a cascade of solutions. Put simply, restoring terrestrial vegetation to a more natural condition should lead to improvements in overall ecosystem health. Pathways to forest health decline were largely a result of vegetation manipulations. Can't those pathways be retraced?

Alan Barta

Wildfires may trigger major debris flows, floods, and erosion that can significantly harm or even destroy small fish populations and other aquatic biota.

Perhaps, but ecological function does not follow directly from structure. Rather, both structure and function emerge from the underlying process (Bradshaw 1996). In other words, how we get where we wish to go may be just as important as where we wish to go. Silvicultural and prescribed fire management tools can lead to the goal of restoring stand structure and composition, but such restoration may have mixed implications for restoring other aspects of failing ecosystems such as fishes and their habitats. ◆

We do not question the need to manage landscapes and forests in ways more consistent with natural patterns of disturbance (*sensu* Attiwill 1994). We suggest the issue is not whether to do something or nothing, but rather to weigh the risks of our actions as objectively as possible. To that end, we believe it is useful to consider some of the physical and biological processes likely to influence streams, fish habitats, and populations after large wildfires or management intended to reduce the probability of such fires.

Physical Processes

The intensity and distribution of natural disturbance and ecosystem responses to disturbance are strongly influenced by landscape attributes. Swanson et al. (1988) suggest that land form position and slope gradient affect flow of energy and matter (water, solutes, and particulate matter) and, therefore, influence the pattern and intensity of disturbance by flow processes. Thus, natural barriers such as ridges and valley bottoms control the spread of fire, and slope and channel gradients dictate the velocity and travel distance of sediment movement. The result is a somewhat predictable, although dynamic, patchiness of disturbance at the landscape scale, constrained by land form and slope gradients. Disturbance due to management activities likely is less constrained because of human ability to circumvent natural barriers. Swanson et al. (1990) suggest that this has led to creation of new landscape disturbance patterns in the Pacific Northwest with little regard for the ecologic design of management activities. The implications of the changed disturbance regime at the landscape scale are not easily interpreted. However, a wealth of information exists about natural and management induced disturbances as they affect physical processes on mountain slopes.

Temperature—Changes in stream temperature are controlled by the amount of energy (heat) exchange and the mass of water in the stream. While many processes affect energy exchange, including evaporative cooling and conductive transfer with substrate, the controlling factor is solar radiation (Brown 1969). There is anecdotal evidence of dramatic, short-term temperature elevations associated with fire in small streams (Minshall and Brock 1991). In addition to latitude, slope, aspect, frequency of cloudy sky, and time of year, incoming radiation is correlated with shading provided by streamside vegetation and terrain and, therefore, is subject to change from fire (Helvey 1972; Amaranthus et al. 1989) or logging (Brown and Krygier 1970; Meehan 1970; Patton 1973). Loss of shading can cause mean annual maximum temperatures to rise by as much as 15°C (Brown and Krygier 1970). The effects of canopy loss on stream temperature by fire or logging disturbance may be approximately equal if the fire consumes the canopy. However, many fires don't consume canopy foliage, and conifer needles may be retained following crown fires for up to a year. In addition, standing tree boles adjacent to streams provide some shade for many years. Streamside vegetation recovery is usually rapid after logging or a fire unless soil is compacted during timber harvest and yarding. Recovery of streamside vegetation is generally faster in moist coastal ecosystems than in drier inland systems, and recovery is faster at lower elevations than higher ones (Beschta et al. 1987).

Erosion—Disturbance by fire, harvest activities, and road construction invariably results in greater erosion and sediment production; however, the severity and longevity of increase is highly dependent on site properties and the kind of disturbance. At the broad scale, erosional processes differ due to lithology, land form, and local climate. Consequently, landscapes are predisposed to variation in erosional processes and efficiencies of sediment transport during times of relative stability and after disturbance. A thorough discussion that distinguishes these process differences in natural landscapes can be found in Swanston (1991) and Megahan (1991).

Disturbances that change the way a slope handles water are likely to result in more erosion and to persist longer than disturbances that reduce cover. Fire consumption of forest floor or removal of soil cover during log skidding generally increases surface erosion rates. Effects are usually short-lived (a few years), although harsh sites may not reestablish effective ground cover for more than a decade (Megahan et al. 1995). In contrast, fire-induced water repellence (DeBano et al. 1970) increases the risk of surface rilling, "slurry flows" (Rinne 1996), debris flows, and hyperconcentrated flood events by reducing the infiltration capacity of soils. While reduced infiltration due to fire-induced water repellence is brief—usually a few years—channel recovery (reestablishing soil and mature vegetation) after debris flows or floods may take decades (Megahan 1991) or even centuries (Benda 1985). These effects are most common in headwater basins, although structure and habitat in receiving streams may be altered for long periods through deposition and scour (Swanston 1991).

Timber harvest and fire both increase the likelihood of mass erosion on sites. The presence of roots and tree stems imparts a measure of strength to soils, decreasing the likelihood of mass failure (Ziemer and Swanston 1977; Gray and Megahan 1981). This is particularly important in colluvial soils because frequent erosion and deposition in these sites provide little time for soil development and pedogenic processes such as weathering and organic matter accretion to promote soil structure and increase resistance to mass erosion. Soil strength declines as roots decay and typically reaches a minimum in 5–10 years after disturbance, but buttressing by large trees may last for several decades after mortality. Vegetation loss also results in decreased water removal from soil by transpiration. Higher antecedent soil moisture during rainfall or snowmelt increases the probability of flooding and the occurrence of shallow, rapid landslides, and can activate flowing water in ephemeral draws. In cohesionless soils, "sapping" failures (Megahan and Bohn 1989) and channel headcutting may greatly accelerate sediment production from ephemeral draws.

Road construction causes the most severe disturbance to soils on slopes, far overshadowing fire and logging as a cause of accelerated erosion (Swanson and Dyrness 1975; Beschta 1978; Reid and Dunne 1984). Perhaps the major reason is that the excavation required to build a road on a slope results in a disruption of subsurface water transport, bringing water to the surface where flow is concentrated, and velocities are much higher (Megahan 1972). Other reasons include disruption of soil structure and vegetation rooting; sidecasting of loose, unconsolidated roadfill; increased connections between roads and streams by gully formation and slides; and oversteepened cutslopes. Accelerated sediment production from roads depends on how recent construction was and on whether road design features that mitigate erosion such as special drainage features, structures, revegetation, and surfacing treatments were incorporated. Acceleration factors for sediment production range from tens to hundreds of times over natural rates in forested areas, based on various studies in the mountain west (Furniss et al. 1991). Typically, sediment production rates decline by an order of magnitude within three years of construction; however, these accelerated rates remain for the life of the road. Secondary surface erosion on mass failures is often chronic and may be as damaging to stream biota as episodic inputs from mass failures because the sediment particles are finer and are delivered throughout a long time.

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Surface and mass-erosion risk are increased after fire or logging, and rates and recovery times are generally similar. Fire-induced water repellence may greatly increase the likelihood of flooding, surface soil rilling, and hyperconcentrated flow events in first- and second-order channels. This elevated risk diminishes rapidly with time, but accelerated erosion and channel scour can be dramatic when high-intensity summer rainstorms hit recent burns if soils are water-repellent. Sediment production from logging and fire accelerates at similar rates. Increases may range from 10- to 100-fold and usually decline to natural levels within a decade (Megahan 1980). Rapid erosion from road disturbance is typically greater than accelerated erosion from fire or logging disturbance and persists for the life of the road. Effective road placement and design features can mitigate erosion from roads, but studies clearly identify accelerated sediment production from roads to be larger and more chronic than sediment production from fire or logging.

Streamflow—Water yield and timing of peak and low flows also are affected by disturbance. Harvesting and fire may alter the spatial pattern of snow deposition and redistribution, and change interception, evaporation, rate of snowmelt, and storage of water in soil by decreasing transpiration. Changes in water yield after disturbance depend on climatic factors, vegetation changes, and soil properties. Typically, the largest increases in yield occur on sites that received precipitation or snowmelt during the growing season, sites that were fully vegetated and had a large change in transpiration, and on deep soils with a high water-holding capacity. In the West, water yield increases after vegetation disturbance range from not detectable to 40% in the first year

(Chamberlin et al. 1991); typical recovery to predisturbance levels take 30–50 years. Peak flows average 20% higher after a forest is cut in small patches in a snowmelt-dominated regime (Troendle and King 1985). Effects of roads on streamflow response are varied. Peak flows and flow duration have either increased, decreased, or remained unchanged in basins after road construction (King and Tennyson 1984; Jones and Grant 1996). Maximum effects on fish habitat (both beneficial and detrimental) probably occur when disturbance alters peak flows or low flows.

Water Chemistry—Stream chemistry changes have been documented after disturbance by fire or logging. Short-term (1-3 *years)* elevated concentrations of dissolved nitrate, cations, and alkalinity have been reported. With the exception of extreme conditions during or immediately following fires across some small streams (Minshall et al. 1989; Rinne 1996), such changes are inconsequential to aquatic biota. Downstream eutrophication, particularly in lakes and reservoirs, may be a concern that we do not consider in this paper.

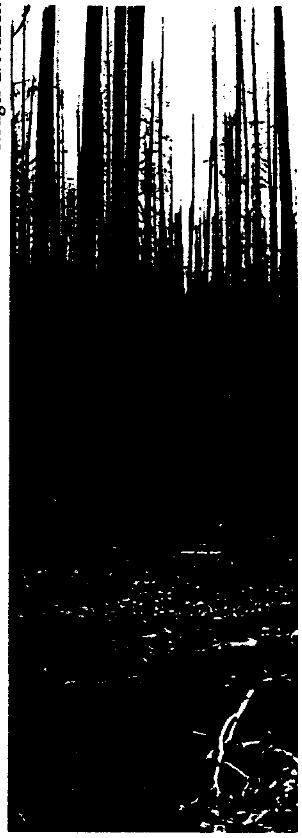
Biological Processes and Fishes

The effects of large fires on fishes are direct and indirect. Direct effects may result from changes in water temperature and chemistry (Minshall et al. 1989; McMahon and de Calesta 1990; Minshall and Brock 1991). Direct effects may include mortality and displacement of individuals, but few studies have documented the extent of these responses (McMahon and de Calesta 1990). Clearly, fires can result in immediate mortalities (McMahon and de Calesta 1990; Minshall and Brock 1991; Rieman et al. 1997), but actual mechanisms and interactions are poorly understood (Minshall et al. 1989; Minshall and Brock 1991; Bozek and Young 1994) and may be highly variable. Rieman et al. (1997) noted areas where no live fish and numerous dead fish were found immediately after an intense fire burned through riparian corridors of two streams in Idaho. In a third stream in a similar setting and immediately following a similar fire, fish densities were depressed in intensely burned reaches, but no areas were devoid of fish. Minshall et al. (1989) suggested that changes in water temperature and water chemistry during or immediately after the Yellowstone fires were relatively minor. Minshall and Brock (1991) suggested such effects were more likely to be important in smaller rather than larger streams. Rinne (1996) found no significant reduction in densities of fish as a direct result of a large fire burning across three headwater streams in Arizona.

Indirect effects of large fires are somewhat better understood and can be dramatic. Bozek and Young (1994) noted mortalities of Yellowstone cutthroat trout (*Oncorhynchus clarki bouvieri)* associated with increased suspended sediments in streams two years after large fires in the Greater Yellowstone ecosystem. Floods and debris flows triggered by fires and subsequent storm events have apparently eliminated fish from stream reaches in Montana (Novak and White 1989) and Idaho. Others have documented the local extinction of small populations of salmonids after similar events (Propst et al. 1992; Rinne 1996).

A general conclusion is that large fires can, in the short term, result in substantial mortality and even local extinctions. However, the result may be largely a function of scale. The relative magnitude of an effect is likely to be strongly influenced by the intensity and severity of the fire, which will, in turn, be influenced by patterns of living vegetation, fuel loads, fire weather, and terrain. Small streams are more likely to be influenced stronger than larger streams. Even when large regions are burned, the mosaic of conditions across streams will produce a patchwork of disturbance effects (e.g., Minshall and Brock 1991). The effects of fires on watershed and ecological processes may be evident for decades (Minshall et al. 1989; Minshall and Brock 1991; Huntington 1995), but they may not necessarily be catastrophic.

Studies of fish populations after large disturbances or experimental defaunation have concluded that population recovery can occur quickly, frequently within a few years (Niemi et al. 1990; Detenbeck et al. 1992), weeks (Sheldon and Meffe 1995), or even days (Peterson and Bayley 1993). To consider risks and tradeoffs associated with large fires and management designed to mitigate those fires, we should consider characteristics of populations and habitats likely to influence biological responses. Two elements seem important: (1) refounding of affected streams or reaches through dispersal from local or internal refugia and (2) stabilizing effects that emerge from complex life histories and overlapping generations in fish populations.



Rodger L. Nelson

Refugia—Rieman et al. (1997) followed responses of three rainbow trout and two bull trout populations in three watersheds after two of the largest, most-intense fires recorded in Boise National Forest, Idaho. Despite the large sizes, the fires in these watersheds left an assortment of intensely burned, lightly burned, and unburned areas that immediately and directly influenced fish distributions. In one stream, depauparate reaches were interspersed with areas supporting high densities of fish. In two streams, moderate or high densities of fish were found immediately below long (2 km–5 km) reaches devoid of fish. Fish may have been completely eliminated from some smaller tributary streams. In years after the fires, debris flows occurred in tributary streams of each watershed. Although populations were depressed, fires in 1992 and 1994 did not produce a uniform or complete elimination of fish or disruption of habitat. Rather, fish numbers increased somewhat dramatically, often exceeding those before the fire or those in streams not influenced by fire (Rieman et al. 1997).

Population responses after the Boise fires were apparently influenced by the presence of refugia (i.e., habitats or stream reaches that supported concentrations of fish during and after the fires). In one system where fish persisted in patches throughout the mainstem, numbers and age structure approached preburn conditions in the first year. Fish were reestablished in depauparate reaches by dispersal from both upstream and downstream sources over short distances. In two other streams where fish were apparently eliminated in the upper reaches and some tributaries, recolonization must have occurred through dispersal from downstream sources over longer distances and time.

The mechanism and rate of recolonization are likely to be influenced by the local environment (Sheldon and Meffe 1995). Mechanisms that stimulate recovery or compensate for habitat losses might even be triggered or enhanced by the disturbance (Bisson et al. 1988, 1997; Minshall et al. 1989). Although some scientists have reported degradation of juvenile habitats after fire (Minshall et al. 1989), such changes also may result in increased production. Rieman et al. (1997) found high densities for young-of-the-year trout in several reaches one and two years after the Boise fires in all of the watersheds directly influenced by the fires. In small, very cold streams, increased solar exposure may result in warmer water, increased primary and secondary production (Minshall et al. 1989; Minshall and Brock 1991), and faster growth or higher carrying capacities for juvenile fish (Murphy and Meehan 1991). Similar positive responses in growth and production have been observed for other salmonids after large disturbances (Bisson et al. 1988). Although fires may kill fish and depress production in some life stages for some time (Minshall et al. 1989), they also may create the potential for important compensation and subsequent dispersal within or among populations. The importance of spatially redundant and complex or heterogeneous habitats to the persistence of populations is well established in theory (den Boer 1968; Dunning et al. 1992), and there is growing empirical support (e.g., Pearsons et al. 1992). A complex, well-dispersed network of habitats is likely to be an important element in the persistence of fishes during and after large fires.

Streamside vegetation grows below the badly burned trees in Fall Creek, Payette National Forest, after a fire during summer 1994.

Life History—Many salmonids exhibit a suite of life history forms that include varied patterns of movement, age and timing of maturity, and habitat utilization (Northcote 1992; Thorpe 1994a, 1994b). Multiple forms (i.e., resident and migratory) may occur in proximity to each other, perhaps even in sympatry. The diversity and plasticity of life histories have been viewed as stabilizing mechanisms for populations in variable environments (Gross 1991; Thorpe 1994 b). For example, Rieman et al. (1997) and Novak and White (1989) provide evidence that salmonid populations, severely depressed by the effects of fires, persisted through the presence of a migratory life history form. In essence, although fires caused severe mortality or even eliminated the fish in some streams, populations persisted because some members had migrated outside of the affected area, returning later to spawn. Life history pattern apparently provided both a temporal and spatial hedge against extinction; had these populations been restricted to nonmigratory forms, they might well be extinct. Similar results have been reported for salmonid populations influenced by other disturbances (Titus and Mosegaard 1992; Armstrong et al. 1994).

Complex life histories may be the result of historical patterns of disturbance. For example, spawning and rearing habitats of bull trout are distributed primarily in colder, higher-elevation watersheds (Rieman and McIntyre 1995). Because high-elevation areas throughout the region were more likely to experience mixed or high-intensity fires (Arno 1980), we might expect that, in an evolutionary sense, bull trout and similarly distributed species are well acquainted with large, intense fires. The existence of complex life histories such as the mixed migratory behaviors and overlapping generations found in the Boise River basin could be the expression of strategies that have emerged because of periodic disturbances like fire.

The broad expression of life histories appears dependent on the heterogeneity and connectivity of habitats. Migratory patterns require access to networks of streams that may extend from hundreds of meters to hundreds of kilometers (Bjornn and Mallet 1964; Fausch and Young 1995). Although different forms may occur in the same streams or watersheds, evidence exists that some differences have emerged as adaptations to, or phenotypic expressions against, a template of heterogeneous habitats (Healey 1994; Healey and Prince 1995; Lichatowich and Mobrand 1995). If the spatial and temporal complexity of habitats is lost, the expression of complex life histories may be lost as well.

The Effects of Management—If refugia and the expression of life histories are critical to the recovery and persistence of fishes influenced by large fires, the condition of available habitats and effects of past management should be key elements influencing evaluations of risk. The potential effects of management on the ability of populations to respond to large disturbances must be weighed. Throughout the Pacific Northwest, the effects of chronic watershed disturbance by road-building and timber harvest; introduced species; and barriers such as dams, diversions, and road culverts have resulted in fragmented, isolated salmonid populations and the elimination or restriction of life-history patterns (Rieman and McIntyre 1993; Frissell et al. 1997). The status of native fishes has been negatively associated with indices of human-related disturbance such as the density of roads (Lee et al. 1997). Although the mechanisms are varied, the risks are clear. Through efforts to intensely manage land and associated activities such as timber harvest, we risk expanding the disruption of watersheds and populations of native fishes. In general, attempts to minimize such effects in any single watershed have led to the dispersal of disruptive activities across broad areas (Reeves et al. 1995). The chronic nature of land management and other human-related disturbances has led to lost spatial complexity of stream environments that ultimately may be reflected in the loss of complexity, diversity, and distribution of populations and life histories (Frissell et al. 1993; Reeves et al. 1995). That loss might well erode the ability of populations to respond to the effects of large fires as well as large storms, floods, and other natural events that we cannot hope to control.

Conclusions and Future Direction

The tradeoffs between vegetation management and wildfire are not simple. In some cases short-term rates of erosion and sediment delivery after a fire may be larger than the effects of roads and timber harvest. After a fire, changes in vegetation and watersheds influencing hydrologic and temperature regimes and erosion may persist for years, perhaps decades. However, the long-term legacy can be important. Substantial inputs of large wood and coarse sediments also are likely to follow large fires (Brown 1989; Reeves et al. 1995; Young 1994). The larger materials often store fine sediments and provide the hydraulic complexity necessary for sorting substrates important to fish habitat. After fires, intense debris flow and scour events that generate sediment are often localized (Megahan 1991) and prevail primarily in smaller, high-gradient channels (Swanson et al. 1990). Although the volumes of fine sediments can be large, they may be relatively short-lived and patchy in relation to the effects of other more-chronic disturbances associated with roads and timber harvest (Reeves et al. 1995). Historically, the episodic contribution of coarse debris may have been key to the creation and maintenance of complex instream habitats (Swanson and Lienkaemper 1978; Reeves et al. 1995). Emerging theory supports the idea that natural disturbances have been critical to the maintenance of such habitats, the productivity of associated populations (Reeves et al. 1995; Bisson et al. 1997), and the broad expression of life histories. Recent fire suppression could well have contributed to the overall decline in productivity of fish habitats throughout the region.

Large fires can produce local extinctions of small, isolated populations. However, many species and populations may still have the ecological diversity necessary to persist. Although wildfires may create important changes in watershed processes often considered harmful for fish or fish habitats, the spatial and temporal nature of disturbance is important. Fire and the associated hydrologic effects can be characterized as "pulsed" disturbances (*sensu* Yount and Niemi 1990) as opposed to the more chronic or "press" effects linked to permanent road networks. Species such as bull trout and redband trout appear to have been well adapted to such pulsed disturbance. The population characteristics that provide for resilience in the face of such events, however, likely depend on large, well-connected, and spatially complex habitats that can be lost through chronic effects of other management. Critical elements to resilience and persistence of many populations for these and similar species will be maintaining and restoring complex habitats across a network of streams and watersheds. Intensive land management could make that a difficult job.

Fire and management-related disturbances may cause important, and in some respects similar, changes in physical and biological systems. However, we do not expect conventional timber harvest and road to produce effects equal to fire. Also clear is that attempts to manipulate the structure, composition, and processes of whole ecosystems are largely experimental (Baker 1994; Stanley 1995; Kimmins 1996). In haste, managers have justified actions by citing such concerns as risks to biological communities and sensitive fishes. Undoubtedly, cases exist where the risks of large fires outweigh the risks of intensive management, but those will be clarified only through careful evaluation. Because past management and human disturbance have led to much change in terrestrial and aquatic ecosystems, we suspect that forested landscapes most in need of restoration will often coincide with watersheds and aquatic systems in similar condition (Quigley et al. 1996). That pattern could represent both potential conflicts and opportunities for managers struggling with issues of forest health, wildfire, and sensitive fishes.

Where important, but depressed and small or isolated populations of sensitive fishes persist in landscapes at high risk of uncharacteristic wildfire, populations are threatened both by our management and the lack of it. In these circumstances management should proceed only with the greatest possible care. Silvicultural prescriptions that do not require new or reconstructed road systems and that emphasize prescribed fire or low-impact logging and yarding systems would be clear priorities. Restoring watershed processes through the stabilization and obliteration of roads, hill slope stabilization, and revegetation of riparian areas could be key elements in any plan. Careful watershed and hydrologic analyses are required to weigh any tradeoffs.

Where populations persist in a matrix of healthy watersheds and productive habitats mixed with those in poorer condition, the opportunity for more progressive and adaptive management is greater. Because healthier populations are more likely to persist in the face of even large fires, the need to restructure forests is unlikely to be an immediate priority from an aquatic perspective and, potentially, is a threat. In many cases the extent of the forest health problem is large, and not all lands can be treated in the foreseeable future, even if tradeoffs were clear. We suggest that logical priorities lie in watersheds that are either less or not important in terms of aquatic biological diversity or critical populations, or that have been so strongly altered through past management that further losses are likely to be minor. Prioritizing activities in these areas offers several advantages. First, because management of large systems is largely experimental, new approaches can be tested without placing key aquatic systems at further risk. Second, successfully reestablishing more natural patterns and processes could lead to long-term restoration of more complex, productive aquatic habitats. The long-term restoration of a matrix of whole watersheds could ultimately lead to the wider distribution of habitats necessary to support a broader expression of aquatic biological diversity. The potential for success might be substantially greater if forest restoration projects included the obliteration of particularly damaging roads. Third, working in the matrix would represent progress toward reducing the continuity of fuels, diminishing the risk of uncharacteristically large fires, and restoring biological diversity in terrestrial systems (*sensu* Carey and Curtis 1996).

The approach we suggest will require a broader perspective. As managers attempt to restore more natural patterns in forests (and aquatic ecosystems), both the distribution and intensity of activities should be guided by a landscape perspective of risk and opportunity. A strategic vision of historical condition and variation, current patterns, and long-term potential will be critical. It is unrealistic to hope to return all landscapes to their historical conditions from either an aquatic or forest perspective. Prioritizing efforts that minimize risks to remaining elements while maximizing learning and opportunities to create something more functional will be key to progress. As managers learn to create more natural patterns and restore critical processes through active manipulations of forests and watersheds, landscape management that incorporates planned disturbances could conceivably replace a system of watershed and riparian reserves as the solution for maintaining aquatic diversity and ecological function in forested ecosystems (Reeves et al. 1995; Carey et al. 1996). Ultimately, we believe that healthy forests and healthy aquatic communities should be elements of the same problem (*sensu* Franklin 1992), not competing issues in resource allocation.

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